



HAL
open science

Seasonal variations of metals and metalloids in atmospheric particulate matter (PM_{2.5}) in the urban megacity Hanoi

Sandrine Chifflet, Léa Guyomarc'H, Pamela Dominutti, Lars-Eric Heimbürger-Boavida, Bernard Angeletti, Pascale Louvat, Jean-Luc Jaffrezo, Cam Tu Vu, Gaelle Uzu, Xavier Mari

► To cite this version:

Sandrine Chifflet, Léa Guyomarc'H, Pamela Dominutti, Lars-Eric Heimbürger-Boavida, Bernard Angeletti, et al.. Seasonal variations of metals and metalloids in atmospheric particulate matter (PM_{2.5}) in the urban megacity Hanoi. *Atmospheric Pollution Research*, 2024, 15 (1), pp.101961. 10.1016/j.apr.2023.101961 . hal-04255510

HAL Id: hal-04255510

<https://hal.science/hal-04255510>

Submitted on 24 Oct 2023

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

1 **Seasonal variations of metals and metalloids in atmospheric particulate matter (PM2.5) in the**
2 **urban megacity Hanoi**

3

4 Sandrine Chifflet^{a*}, Léa Guyomarc'h^a, Pamela Dominutti^b, Lars-Eric Heimbürger-Boavida^a, Bernard
5 Angeletti^c, Pascale Louvat^d, Jean-Luc Jaffrezo^b, Cam Tu Vu^e, Gaëlle Uzu^b, Xavier Mari^{a,e,g}

6

7 ^a Aix Marseille University, Université de Toulon, CNRS, IRD, MIO (UM110), 13288 Marseille, France

8 ^b University of Grenoble Alpes, CNRS, IRD, INP-G, IGE (UMR 5001), 38000 Grenoble, France

9 ^c Aix Marseille University, CNRS, IRD, Collège de France, INRAE, CEREGE, 13545 Aix-en-Provence
10 Cedex 4, France

11 ^b Université de Pau et des Pays de l'Adour, E2S UPPA, CNRS, IPREM, UMR 5254, Pau, France

12 ^e Water-Environment-Oceanography (WEO) Department, University of Science and Technology of
13 Hanoi (USTH), Vietnam Academy of Science and Technology (VAST), 18 Hoang Quoc Viet, Hanoi,
14 Vietnam

15 ^g IRD, Chulalongkorn University, 254 Henri Dunant Road, Pathumwan, 10330 Bangkok.

16

17 * Corresponding author: sandrine.chifflet@mio.osupytheas.fr; Phone: + 33 4 86 09 05 32;

18

19 For submission to Atmospheric Pollution Research

20

21 September 2023 (revised version)

22 **Abstract**

23 Fine particulate matter (PM2.5) in the atmosphere is of particular concern due to its adverse
24 effects on human health and its impact on global warming. southeast Asia is a hot spot for fossil fuel
25 combustion with recurrent release of large plumes spreading over the ocean and neighbouring
26 countries. Due to the complex mixture of PM2.5, the atmospheric sources contribution related to
27 local and regional emissions in Hanoi (northern Vietnam) is still ill-constrained. Here, we present a
28 year-round study (November 2019 to December 2020) with measurements of 18 metals and
29 metalloids (MM) and lead isotopes in the PM2.5 fraction to quantify weather-related atmospheric
30 inputs and to assess risk to human health. Anthropogenic inputs from fossil fuel combustion
31 accounted for about 80% in PM2.5. We found high PM2.5-bound MM concentrations often
32 exceeding national and global standards with a low risk of chronic inhalation and carcinogenicity,
33 mainly attributable to Cr. During winter monsoon (northeastern winds), stable weather conditions
34 led to the enrichment of long-range air mass transport of local particulate emissions. During the
35 summer monsoon (southeastern winds), warm and moist winds reduced coal contribution in PM2.5.
36 Our study highlights the need for a strict implementation of policies to control hazardous MM
37 emissions by reducing fossil fuel combustion. On the one hand, reducing coal-related activities could
38 reduce Cr emissions and therefore improve the risks to human health. On the other hand, public
39 policies should encourage conversion to green transport in order to reduce petrol combustion and
40 thus limit global warming.

41

42

43 Keywords: contamination, mercury, lead isotopes, atmospheric transport, fossil fuel combustion

44

45

46

47 **1. Introduction**

48 Atmospheric particulate matter (PM) alters the climate (Fuzzi et al., 2015) and has a negative
49 effect on human health (Janssen et al., 2011; Pope et al., 2011; Xing et al., 2016). Fine particles
50 smaller than 2.5 μm (PM_{2.5}) can penetrate deep into the lungs and cause asthma, lung disease,
51 heart disease and cancer, depending on the amount inhaled and the time of exposure and the
52 susceptibility of the person. Globally, inhalation of atmospheric PM is currently estimated to cause
53 3.3 million premature deaths per year worldwide, and could double by 2050 (Stanaway et al., 2018;
54 Lelieveld et al., 2020). Atmospheric PM is a complex mixture of various compounds such as organic
55 and elemental carbon, sugars, polycyclic aromatic hydrocarbons, secondary inorganic aerosol (*e.g.*,
56 SO_4^{2-} , NH_4^+ , NO_3^-), *etc.* (Finlayson-Pitts et al., 2020).

57 Among these compounds, metals and metalloids (MM) are of particular concern for human
58 health (Palleschi et al., 2018; Rafiee et al., 2020; Soetrisno and Delgado-Saborit, 2020). In small
59 amounts, MM are essential to maintain biological functions, but in larger amounts they can damage
60 cellular functions and cause toxicity in certain organs such as kidneys, brain, blood, skin, heart (Mitra
61 et al., 2022). Due to their chemical coordination and oxidation-reduction properties, at high
62 concentrations, MM can compete with each other. One element can bind to protein sites in place of
63 another, causing DNA damage and perturbing protein function and enzyme activity (Balali-Mood et
64 al., 2021; Witkowska et al., 2021; Jaishankar et al., 2014). Among them, As, Cd, Cr are of greatest
65 concern due to their numerous sources of emissions, high toxicity, and persistence in urban areas (Li
66 et al., 2021).

67 Atmospheric PM is ubiquitous and can originate from natural and anthropogenic sources
68 (Pacyna and Pacyna, 2001; Gieré and Querol, 2010). PM-bound MM can be naturally introduced into
69 the atmosphere through wind erosion of soil or sea surface, volcanic emissions and forest fires,
70 mainly as coarse fraction ($> 3\text{-}4 \mu\text{m}$) and relatively non-solvable (Li et al., 2009; Bacon et al., 2011). By
71 contrast, MM from anthropogenic activities can be introduced into the atmosphere through many
72 pathways such as fossil fuel combustion, mining, ferrous and non-ferrous metal manufacturing,
73 industrial production (*e.g.*, cement, ceramic, glass, fertilizers), road traffic, waste incineration,

74 domestic heating etc. (Pacyna et al., 2007; Hao et al., 2018). Furthermore, anthropogenic MM tend
75 to be associated with fine particles (Hein et al., 2022) and far exceed natural concentrations in
76 heavily urbanized/industrialized areas (McNeill et al., 2020; Guo et al., 2023).

77 Atmospheric PM is a global concern and even more so for densely populated Asian countries
78 (Xu et al., 2020; Zhang et al., 2021; Sakunkoo et al., 2022). Vietnam's capital of Hanoi experienced
79 rapid urbanization over the last decades, resulting in a severe increase in PM_{2.5} atmospheric load
80 with an average annual concentration greater than 45 $\mu\text{g m}^{-3}$ (Makkoken et al., 2023), well above
81 national (25 $\mu\text{g m}^{-3}$, QCVN 05:2013/BTNMT) and updated global guidelines (5 $\mu\text{g m}^{-3}$; WHO, 2021).
82 During peak hours, congested traffic significantly increases fine particulate emissions, contributing up
83 to 40% of PM_{2.5} (Cohen et al., 2010; Quang et al., 2021). Northern Vietnam is heavily industrialized
84 with coal-fired power plants as the main source of energy (Huy and Kim Oanh, 2017; Roy et al.,
85 2021). Furthermore, agricultural burning is widely used in rural and suburban areas (Lasko et al.,
86 2018; Le et al., 2020). In addition, long-range air mass transport from China to northern Vietnam
87 contribute to the increase in PM_{2.5}, with concentrations regularly exceeding 150 $\mu\text{g m}^{-3}$ during the
88 northeast winter monsoon (Ly et al., 2021; Ngoc et al., 2021; Makkoken et al., 2023).

89 Although PM_{2.5}-bound MM are strongly studied in China (e.g., Liao et al., 2023; Zhou et al.,
90 2023; Guo et al., 2022; Duan et al., 2021), the latter are poorly documented in Vietnam. In 2020,
91 mean annual concentrations for As, Cd, Cr in PM_{2.5} were found at 6.2, 1.9, 9.4 ng m^{-3} , respectively
92 (Nguyen et al., 2022), exceeding global guidelines for As and Cr (6.0 and 0.25 ng m^{-3} , respectively;
93 WHO, 2021). In addition, a strong variability in PM_{2.5}-bound MM can be observed between studies.
94 In Hanoi, over the same sampling periods (2001 - 2008 and 2002 - 2005), the mean Pb concentration
95 in PM_{2.5} was assessed to 236 ng m^{-3} (Cohen et al., 2010) and 137 ng m^{-3} (Hopke et al., 2008),
96 respectively, namely a difference greater than 50% between the two studies. In addition, maximum
97 measured values were three times lower in 2001 - 2008 than in 2002 - 2005 (5.4 and 15.0 $\mu\text{g m}^{-3}$,
98 respectively), but these results are well above the WHO guidelines (0.15 $\mu\text{g m}^{-3}$; WHO, 2021) for the
99 two studies.

100 Principal Component Analysis (PCA) is a statistical technique widely used to identify the
101 sources of MM, but the correlation matrices are poorly weighted and do not allow them to be
102 quantified. Alternative techniques using Chemical Mass Balance (CMB) or Positive Matrix
103 Factorization (PMF) are mathematical techniques used to solve systems of linear equations. The CMB
104 model requires a large number of samples and the identification of sources is based on the chemical
105 profiles from data obtained mainly in northern America, which may be different for southeast Asia
106 (Henry et al., 1984, Paatero and Tapper, 1994; Pant and Harrison, 2012). With the PMF model, results
107 are constrained to provide positive source contributions, and the uncertainty weighted difference
108 between the observed and predicted species concentration is minimized. However, analyses of large
109 datasets are often time-consuming and use up all the computer's available memory, which slow
110 down the programme and can even cause errors (Hopke, 2000). Using PMF model, Cohen et al.
111 (2010) found that the main sources of PM_{2.5} in Hanoi were from traffic (40 ± 10%), industry
112 (19 ± 8%), and coal (17 ± 7%). The contribution of sources related to local and regional PM_{2.5}
113 emissions is still under debate (Nguyen et al., 2020). In a recent study, Makkonen et al. (2023), found
114 that PM_{2.5} in Hanoi are from local and regional secondary inorganic emissions (18 and 25%,
115 respectively), biomass burning (19%), dust (17%), traffic (12%) and industry (9%). Given the high
116 concentrations of PM_{2.5} and associated MM already observed, it is important to better define and
117 assess anthropogenic contributions in order to limit the risk of exposure to human health. (Nguyen et
118 al., 2022; Vo et al., 2022).

119 Isotopic analyses are commonly used for the identification and quantification of MM sources
120 (Chifflet et al., 2018). Although the stable radiogenic lead isotopic composition (²⁰⁶Pb, ²⁰⁷Pb and ²⁰⁸Pb)
121 can be naturally transferred in the environment *via* weathering processes (Stacey and Kramers,
122 1975), these isotopes are not significantly affected by physico-chemical fractionation processes after
123 mixing with secondary Pb sources thus providing an efficient tool for determining the origins and
124 pathways of Pb pollution (Komárek et al., 2008). In order to better assess the impact of
125 anthropogenic activities on air quality, we are presenting here a year-round study (November 2019

126 to December 2020) with quantitative measurements of 18 MM (Ag, Al, As, Cd, Co, Cr, Cu, Fe, Hg, Mn,
127 Ni, Pb, Sb, Sn, Ti, U, V and Zn) and lead isotopes analyses (^{206}Pb , ^{207}Pb and ^{208}Pb) in PM_{2.5} collected in
128 Hanoi. The objectives are i) to identify and quantify the main anthropogenic sources, ii) to study the
129 impact of weather conditions on MM variability in PM_{2.5} and, iii) to assess potential inhalation risks
130 to human health per MM. Given the energy demand in Southeast Asia and the population density of
131 Hanoi, the objective of this article is not to conduct an exhaustive study of PM_{2.5} sources but to
132 focus on the impact of fossil fuels combustion on air quality.

133

134

135 **2. Materials and Methods**

136 *2.1 Study site*

137 Hanoi is a megacity covering an area of 3,359 km² with a population of 5.2 million in 2023
138 ([Makkoken et al., 2023](#)). The city is located in the Red River delta (northern Vietnam) about 100 km
139 from the Gulf of Tonkin (Fig. 1). The subtropical climate is mainly influenced by the northeast
140 monsoon in winter (November to March) and the southeast monsoon in summer (May to
141 September) with two short intermediate seasons in spring and fall ([Ngu and Hieu, 2004](#)). In winter,
142 the high atmospheric pressure above central China brings cold and calm air masses to northern
143 Vietnam, the low boundary mixing layer favouring the accumulation of local atmospheric particles
144 ([Hai and Kim Oanh, 2013](#)). In summer, the high atmospheric pressure moves toward the southern
145 hemisphere bringing wet and warm air masses to Hanoi ([Hien et al., 2002](#)).

146 Hanoi is surrounded by many suburban industrial activities (*e.g.*, Thang Long industrial Park
147 in the North, Saidong industrial zone in the East, Ngoc Hoi industrial zone in the South) and several
148 highly industrialized provinces in northern Vietnam (Thai Nguyen, Quang Ninh, Bac Ninh, and Hung
149 Yen) ([Hien et al., 2020](#)). Approximately 10 coal-fired power plants are located within 200 km of Hanoi
150 and emit large amounts of PM into the atmosphere
151 (<https://globalenergymonitor.org/projects/global-coal-plant-tracker/tracker/>). In addition, the

152 vehicle traffic increases sharply together with the expansion of urban and industrial activities. In
153 2019, more than 6.6 million vehicles were registered in Hanoi (0.7 million cars and 5.8 million
154 motorcycles) causing serious congestion problems on some roads during peak hours (Huu and Ngoc,
155 2021).

156

157 *2.2 Sampling*

158 PM_{2.5} were collected on the roof top of the University of Science and Technology of Hanoi
159 (USTH; 21.0489°N, 105.8011°E) using a Staplex High Volume Air sampler. The sampling was
160 conducted from November 2019 to December 2020 (14 months) with quartz fiber filters (Staplex
161 Type TFAQ810 of 20x25 cm). PM_{2.5} samples (n = 27) were collected weekly over 24-hour periods (1
162 sample per week), except during the lockdown in April 2020 due to COVID-19 pandemic
163 (corresponding to 4 “missing” samples as of 31/03/2020, 07/04/2020, 14/04/2020 and 21/04/2020).
164 The average air volume filtered –calculated using the recorded air flow and the sampling time– was
165 $1803 \pm 81 \text{ m}^3 \text{ filter}^{-1}$. Filters were pre-combusted (550°C, 12h) before sampling to prevent organic
166 matter and mercury residue (US EPA 2013). Field blanks were handled identically to the PM_{2.5}
167 sampling, but were run on the Staplex sampler for only 1 min (Pekney and Davidson, 2005). The
168 quartz filters were kept frozen in clean aluminium foils and plastic bags until laboratory analyses.

169

170 *2.3 Weather conditions*

171 During the study, weather data (wind speed and direction) were downloaded from the
172 National Oceanic and Atmospheric Administration (NOAA, USA) station at Noibai International
173 Airport (21.2212°N, 105.8072°E), 12 m above ground level. The NOAA station was located 23 km
174 North of the USTH. Local precipitation was downloaded online ([https://www.historique-
175 meteo.net/asia-sud-est/vietnam/2020/05/](https://www.historique-meteo.net/asia-sud-est/vietnam/2020/05/)).

176 For each sampling period, the NOAA’s Hybrid Single-Particle Lagrangian Integrated Trajectory
177 model (NOAA HYSPLIT; Stein et al., 2015; Rolph et al., 2017) was used to draw 24 h backward air

178 mass trajectories based on weather data in the Global Data Assimilation System (GDAS) archive
179 starting at 8 am (01:00 UTC) with a temporal resolution of 1 hour at the sampling site and the height
180 of 500 m (Fig. S1). However, the accuracy of an individual trajectory is limited by various
181 uncertainties, leading to coarse approximation of air mass origin (Draxler et al., 2022). Large numbers
182 of trajectories were statistically used to identify homogeneous groups of transport pathways.
183 Weather conditions (wind roses and trajectory frequencies) grouped by seasonal variations (winter
184 and summer monsoon, intermediate spring and fall) are presented in Fig. 2 using the HYSPLIT model
185 available online (<https://www.ready.noaa.gov/HYSPLIT.php>).

186

187 *2.4 Laboratory analyses*

188 *2.4.1 Metals and metalloids (MM)*

189 Sample processing was carried out in a clean laboratory (ISO 5) at the MIO (Aix Marseille
190 University), using bi-distilled (HCl and HNO₃) and commercial grade (HF; Optima grade, Fisher
191 Chemical) acids as well as ultrapure water (Milli-Q Integral 3, Millipore). The perfluoroalkoxy vials
192 (PFA vials, Savillex) were pre-cleaned in two acid baths (HCl 10%, 90°C, 24 h and HNO₃ 10%, 90°C,
193 24 h). The analytical tubes (metal-free centrifuge tubes, VWR) were pre-cleaned in HCl (10%)
194 overnight at room temperature. PFA vials and analytical tubes were thoroughly rinsed then dried
195 under a laminar flow hood (ISO 1) before using. Sub-samples (47 mm punch diameter) were digested
196 in PFA vials in a mixture of pure acids (HNO₃, 2 mL; HCl, 2 mL; HF, 1 mL), heated at 120°C during 24 h,
197 evaporated to near dryness, and re-dissolved into 3 mL of HNO₃ (2% v/v). The digested samples were
198 then diluted in analytical tubes using HNO₃ (2% v/v) before running MM analyses (modified method
199 3050B; US EPA, 1996).

200 MM (Ag, Al, As, Cd, Co, Cr, Cu, Fe, Mn, Ni, Pb, Sb, Sn, Ti, U, V and Zn) analyses were
201 performed using an inductively-coupled plasma mass spectrometer (ICP-MS NexION 300; Perkin
202 Elmer) at the LA-ICP-MS platform (CEREGE, Aix Marseille University). Indium was used as internal
203 standard to correct for instrumental mass bias. The digestion procedure was assessed using two

204 certified samples, from marine sediments (MESS4, National Research Council of Canada), and from
205 atmospheric PM (NIST1649b, National Institute of Standard and Technology, USA). Our
206 determinations of MM concentrations in certified samples were within the satisfactory target
207 recovery of $100 \pm 15\%$ (Chen and Ma, 2001). All MM concentrations in PM_{2.5} were blank corrected.
208 The latter had values below 1% for most MM, except for Al, Cr, Ni and U, which had values of
209 approximately 3%. Quality control details are presented in Table S1.

210

211 2.4.2 Mercury

212 For total Hg determination, sub-samples (47 mm punch diameter) were analysed by atomic
213 absorption spectrometry after thermal calcination (AMA-254, Leco). The method detection limit,
214 estimated as three times the standard deviation of blank samples, was 20 pg. The certified reference
215 material (MESS4, National Research Council of Canada) was run several times per analytical batch
216 and systematically before starting the measurements to check the accuracy of the measurements.
217 The measured values were always within $\pm 5\%$ of the recommended values.

218

219 2.4.3 Lead isotopes

220 Lead isotopic analyses (^{206}Pb , ^{207}Pb , ^{208}Pb) were performed using a multi collector inductively
221 coupled plasma mass spectrometer (MC-ICP-MS; Nu Instrument) at the I3 platform (IPREM,
222 University of Pau and Pays de l'Adour). All sample preparations were carried out in clean laboratories
223 (MIO, Aix Marseille University). Acids used were either bi-distilled (HCl and HNO₃) or commercial
224 (HBr; Optima, Fisher Chemical). In order to minimise matrix effects during analysis, Pb was extracted
225 from digested samples (see section 2.4.1) and purified by an ion exchange resin (Dowex 1X8, 100-200
226 mesh, Acros Organics) according to a conventional protocol (Monna et al., 1997; Ortega et al., 2012).
227 The purified Pb samples were introduced into the MC-ICP-MS as dry aerosol (*via* an APEX nebulizer
228 attached to a SPIRO membrane; ESI) at $50 \mu\text{g L}^{-1}$. An internal Tl isotopic standard (NIST997; National
229 Institute of Standard and Technology, USA) was added at $12.5 \mu\text{g L}^{-1}$ to all samples to correct mass

230 fractionation effects. In addition, a standard bracketing method was applied to correct the
231 instrumental drift with a Pb isotopic standard (NIST981; National Institute of Standard and
232 Technology, USA). Blank and Pb standard (NIST981) were run several times per analytical batch for
233 the quality control procedure. The averages of the measured $^{207}\text{Pb}/^{206}\text{Pb}$ and $^{208}\text{Pb}/^{206}\text{Pb}$ ratios
234 ($n = 70$) in these standards were 0.9144 ± 0.0002 and 2.1662 ± 0.0011 , respectively, matching with
235 the respective reference values of 0.9146 ± 0.0003 and 2.1681 ± 0.0008 .

236

237 *2.5 Data processing*

238 *2.5.1 Reconstructed mass*

239 PM2.5 was estimated using mass reconstruction equations (Querol et al., 2001; Putaud et al.,
240 2010; Chow et al., 2015). Briefly, the concentration of PM2.5 is expressed as the sum of its
241 representative chemical components, including: inorganic major ions (NH_4^+ , NO_3^- , SO_4^{2-}); organic
242 carbon (OC); elementary carbon (EC, also known as black carbon or BC); geological minerals (Al, Ca,
243 Fe, K, Mg, Mn, Ti, P); sea salts (Na^+) and other elements (Cu, Ni, Pb, V, Zn). Chemical analyses were
244 performed at the Air-O-Sol analytical platform (IGE, Grenoble Alpes University). Details of the PM2.5
245 mass reconstruction are presented in supplementary information.

246

247 *2.5.2 Sources apportionment of Pb*

248 In Vietnam, Pb could come mainly from natural (soil) and anthropogenic (fossil fuel
249 combustion) sources. Therefore, the contribution of each source could be assessed using Pb isotopic
250 ratios ($^{207}\text{Pb}/^{206}\text{Pb}$ and $^{208}\text{Pb}/^{206}\text{Pb}$) with the following linear mixing model (Monna et al., 1997;
251 Chifflet et al., 2018):

$$252 \left(\frac{^{207}\text{Pb}}{^{206}\text{Pb}} \right)_{\text{PM2.5}} = x_1 \left(\frac{^{207}\text{Pb}}{^{206}\text{Pb}} \right)_1 + x_2 \left(\frac{^{207}\text{Pb}}{^{206}\text{Pb}} \right)_2 + x_3 \left(\frac{^{207}\text{Pb}}{^{206}\text{Pb}} \right)_3 \quad (1)$$

$$253 \left(\frac{^{208}\text{Pb}}{^{206}\text{Pb}} \right)_{\text{PM2.5}} = x_1 \left(\frac{^{208}\text{Pb}}{^{206}\text{Pb}} \right)_1 + x_2 \left(\frac{^{208}\text{Pb}}{^{206}\text{Pb}} \right)_2 + x_3 \left(\frac{^{208}\text{Pb}}{^{206}\text{Pb}} \right)_3 \quad (2)$$

$$254 x_1 + x_2 + x_3 = 1 \quad (3)$$

255 where the numbers in subscript (1, 2 and 3) represent sources, and x their relative contribution.

256

257 2.5.3 Health risk assessment

258 Health risk assessment provides indexes of adverse health effects for a specific group of
259 population over a specific period (US EPA, 2005). It can be expressed as hazard quotient to assess the
260 non-carcinogenic risk (HQ) and carcinogenic risk (CR). Some MM involved in carcinogenic processes
261 were classified by the International Agency for Research on Cancer (IARC, 2023) as hazardous
262 carcinogens (Group 1: As, Cd, and Cr^{IV}), probably carcinogens (Group 2A: Pb and Co^{0,II}), carcinogens
263 (Group 2B: Ni, Ti, and V), pollutants of potential interest (Group 3: Hg⁰, Mn, Co^{III}, and Cr^{0,III}).

264 The inhalation exposure concentration (EC, $\mu\text{g m}^{-3}$) was expressed as follows (US EPA, 2009):

$$265 \quad EC_i = \frac{C_i^{95\%} \times ET \times EF \times ED}{AT_n} \quad (4)$$

266 where $C_i^{95\%}$ is the mean concentration (95% upper confidence limit) of a metal or metalloid in PM2.5
267 ($\mu\text{g m}^{-3}$); ET is the exposure time (24 h d^{-1}); EF is the exposure frequency (365 d yr^{-1}); ED is the
268 exposure duration (yr); and AT_n is the averaging time ($= ED \times 365 \text{ d} \times 24 \text{ h d}^{-1}$ and $70 \text{ yr} \times 365 \text{ d yr}^{-1}$
269 $\times 24 \text{ h}$ for non-carcinogens and carcinogens, respectively). The reference values (ET , EF , ED and AT_n)
270 are from the EPA Integrated Risk Assessment System (EPA IRIS) website (<https://epa.gov/iris>).

271 The mean concentration of a metal or metalloid in sample ($C_i^{95\%}$) is considered as the
272 reasonable maximum exposure and was calculated as follows (Zhang et al., 2021):

$$273 \quad C_i^{95\%} = \exp\left(\overline{\ln C_i} + 0.5 \times S_{\ln C_i}^2 + \frac{S_{\ln C_i} \times H_{0.95}}{\sqrt{n-1}}\right) \quad (5)$$

274 where $\overline{\ln C_i}$ and $S_{\ln C_i}$ are the arithmetic mean and standard deviation of the log-transformed C_i ,
275 respectively; n is the number of samples. Statistical $H_{0.95}$ values that depend on $S_{\ln C_i}$, n , and the
276 confidence level (0.05) are taken from Gilbert (1987).

277 The non-carcinogenic (HQ) and carcinogenic (CR) risks associated with chemical inhalation
278 exposure were calculated as follows (US EPA, 2009):

$$279 \quad HQ_i^{inh} = \frac{EC_i}{RfC_i \times 1000} \quad (6)$$

280
$$CR_i^{inh} = EC_i \times IUR_i \quad (7)$$

281 where RfC_i and IUR_i are the reference concentration and the inhalation unit risk for a metal or
282 metalloid i , respectively. $HQ_i > 1$ indicates a potential chronic effect through inhalation of a metal or
283 metalloid i . CR_i indicates the probability that a person will develop cancer by inhaling certain MM
284 with varying risk of very low ($CR \leq 10^{-6}$), low ($10^{-6} \leq CR < 10^{-4}$), moderate ($10^{-4} \leq CR < 10^{-3}$), high
285 ($10^{-3} \leq CR < 10^{-1}$), very high ($CR \geq 10^{-1}$) (Hu et al., 2012; Roy et al., 2019; Behrooz et al., 2021). To
286 assess the risk of exposure to multiple MM, the cumulative HI and TCR indexes were calculated as
287 the sum of individual HQ_i and CR_i , respectively. Updated reference values (RfC_i and IUR_i) can be found
288 on the Office of Environmental Health Hazard Assessment website (<https://dtsc.ca.gov>; OEHHA,
289 2019) or in Zhang et al. (2021) and shown in Table S2.

290

291 *2.6 Multivariate statistical analyses*

292 Statistical analyses were performed using Xlstat software package version 2022.2.1
293 (Addinsoft 2023, Boston, USA, <https://www.xlstat.com>). First, the normality and homoscedasticity of
294 data were tested. Then, Principal Component Analysis (PCA) and Hierarchical Cluster Analyses (HCA)
295 were applied to explain the temporal variability of MM in PM2.5. The Pearson correlation matrix was
296 used to examine the relationships between concentrations of the various MM. The Ward's method
297 with Euclidean distances was used to assess similarity or dissimilarity between samples. All statistical
298 tests were performed with a probability (p) < 0.05.

299

300

301 **3. Results**

302 *3.1 Temporal variations of PM2.5 and MM concentrations*

303 PM2.5 concentrations varied from 8.3 to 148 $\mu\text{g m}^{-3}$ during the sampling period (November
304 2019 to December 2020) (Fig. 3; Table S3). High concentrations were observed mainly during the
305 winter monsoon, dominated by east and northeast winds. Conversely, low PM2.5 concentrations

306 were observed during the summer monsoon, dominated by southeast or southwest winds. This
307 monitoring was consistent with recent studies of PM_{2.5} levels in Hanoi (Ngoc et al., 2021, Ly et al.,
308 2021; Bui et al., 2022; Voung et al., 2023). The average annual concentration ($40.2 \pm 26.3 \mu\text{g m}^{-3}$) was
309 1.6 and 8.0 times higher than the national standard ($25 \mu\text{g m}^{-3}$; QCVN 05:2013/BTNMT) and the
310 World Health Organization guidelines ($5 \mu\text{g m}^{-3}$; WHO, 2021), respectively. For the 24-hour exposure
311 to PM_{2.5}, 31.6% and 89.5% of the samples had concentrations above the national standard ($50 \mu\text{g m}^{-3}$;
312 QCVN 05:2013/BTNMT) and the WHO guidelines ($15 \mu\text{g m}^{-3}$; WHO, 2021), respectively. Although
313 very low precipitation (near 0 mm d^{-1}) was recorded in winter and spring, a large increase in
314 precipitation was observed from summer onwards, reaching a maximum of 135 mm d^{-1} in October
315 2020. Wind speed was variable ($2.81 \pm 0.85 \text{ m s}^{-1}$) during the sampling period but the lowest values
316 were recorded in winter.

317 Statistical data (average, minimum and maximum) for PM_{2.5}-bound MM concentrations over
318 the sampling period are presented in Table 1 and detailed values per sample in Table S4. PM_{2.5}-
319 bound MM average concentrations varied over several orders of magnitude, with Al, Fe and Zn being
320 the most abundant (higher than 300 ng m^{-3}), followed by Cu, Mn, Pb and Ti (between 30 and 60 ng m^{-3} ;
321 As, Cd, Cr, Ni Sb, Sn and V (between 1 and 5 ng m^{-3}), and other MM (Ag, Co, Hg and U) at less than
322 0.3 ng m^{-3} . These results are in good agreement with previous studies in Hanoi (Nguyen et al., 2022;
323 Bui et al., 2022; Vuong et al., 2023). No information was found to compare these values with the
324 recommendations of national or global health organizations. However, for PM₁₀ (atmospheric
325 particulate matter < $10\mu\text{m}$ in aerodynamic diameter), the European Community and the WHO
326 recommend limiting annual exposure to Pb, Ni, As, Cd and Cr^{VI} to 150, 20, 6, 5, and 0.25 ng m^{-3} ,
327 respectively (Directives 2008/50/EC; 2004/107/EC; WHO, 2021). In the present study, some targeted
328 MM (Pb, As, Cd and Cr) had 24-hour maximum concentrations above the annual exposure guidelines.

329

330 *3.2 Principal Component Analysis and clustering*

331 The Pearson correlation matrix and Ward's Euclidean distances were examined to highlight
332 the links between PM2.5-bound MM and weather data (wind speed, precipitation and wind direction
333 as qualitative values). High and significant correlations ($r > 0.6$ and $p < 0.05$, $n = 57$) were observed
334 between all variables except Ag, Cu and Precipitation (Table S5). Although wet deposition is
335 considered a major process for removing particulate-bound MM, transfer mechanisms depend on
336 particle size and metal solubility (Pan and Wang 2014; Jaffrezo and Colin, 1988). PM10 is more
337 readily recovered than PM2.5 (Cheng et al., 2021; Zhou et al., 2021) and only the soluble fraction of
338 MM is cleaned from particles (Vithanage et al., 2022). These considerations may explain the lack of
339 correlation between PM2.5-bound MM and precipitation. In addition, the 24-hour sampling
340 conducted on 10/12/2019 and 12/05/2020 made high contribution (26 and 40%, respectively) to
341 factors loadings focusing multivariate statistical analyses on these two specific events instead of the
342 overall study. These two events were characterized by maximum concentrations of PM2.5, Al, Ti, Fe,
343 Co, Pb and Hg on 10/12/2019 and, a low concentration of PM2.5 associated with high concentrations
344 of Ni and Cr on 12/05/2020.

345 To better assess the impact of seasonal variations PM2.5-bound MM, multivariate statistical
346 analyses were reviewed without these two samples, and by redefining some variables (wind speed
347 and direction, precipitation, Ag and Cu) as illustrative parameters (Fig. 4). The original variables were
348 reduced to two factors (F1 and F2) explaining 65% of the total variance with eigenvalues > 1 . F1 had a
349 high positive loading for all MM and negative loading for weather data (wind speed and
350 precipitation). F2 had a high positive loading for Al, Cr, Co, Fe, Ni, Ti, U, and V and a negative loading
351 for As, Cd, Hg, Mn, Pb, Sb, Sn and Zn. The scatter plot corresponding to samples showed two clusters
352 (C1 and C2) clearly distributed along F1 axis with a positive barycentre for C2 and a negative
353 barycentre for C1. Although PM2.5 in C1 could have originated from different air mass trajectories,
354 PM2.5 in C2 all originated from northerly or north-easterly winds (Table S6).

355

356 *3.3 Lead isotopes*

357 The complex mixture of particles does not allow for accurate quantification of anthropogenic
358 sources using statistical studies alone. Due to quantum theory, isotopes are powerful tools for
359 tracing Pb sources and atmospheric transport pathways. Although the isotopic composition of stable
360 radiogenic lead (^{206}Pb , ^{207}Pb and ^{208}Pb) is naturally transferred in the environment (Bollhöfer and
361 Rosman 2000), this composition may vary after mixing with secondary sources emitted by
362 anthropogenic activities (Zheng et al., 2004; Komárek et al., 2008). Despite the phase-out of leaded
363 gasoline in southeast Asia over the past two decades, anthropogenic Pb inputs are still large enough
364 to accurately quantify its sources of emissions (Mukai et al., 2001a; Bing-Quan et al., 2001; Yao et al.,
365 2015). Lead isotopic ratios in PM_{2.5} collected during the sampling period ranged from 0.8005 to
366 0.8702 for $^{207}\text{Pb}/^{206}\text{Pb}$ and from 1.9754 to 2.1149 for $^{208}\text{Pb}/^{206}\text{Pb}$ (Table S7). These ratios were
367 consistent with those found locally in the Tay Ho urban lake in Hanoi (Kikuchi et al., 2010) or with
368 PM₁₀ collected in Haiphong (Chifflet et al., 2018) showing the good relevance of the results.

369

370

371 4. Discussion

372 4.1 Sources identification

373 Atmospheric particles are highly dynamic and reactive compounds whose chemical
374 composition is attributed to a wide range of natural and anthropogenic sources (Laskin et al., 2019).
375 Because PM-bound MM depend primarily on soil composition, raw material use, and primary energy
376 sources (from oil and coal combustion), elemental ratios are widely used to identify emission sources
377 (e.g., Okuda et al., 2004; Hopke and Cohen, 2011). For example, V/Ni is well-known as an indicator of
378 oil combustion (Pacyna and Pacyna, 2001). Previous studies have reported typical V/Ni ratios in heavy
379 oil combustion for ship engines (with values between 3 – 4, Mazzei et al., 2008), in oil combustion for
380 household heating (2.4; Lee et al., 2000) or in road traffic (1.49; Thorpe and Harrison, 2008). In
381 addition, low V/Ni ratios (0.5) have also been reported in Chinese megacities (Beijing, Changchun and
382 Chengdu) and attributed to PM from traffic and/or coal mining activities (Xu et al., 2020). Finally,

383 V/Ni ratios of 0.55 and 0.15 have also been found in Vietnamese coal and diesel, respectively (Chifflet
384 et al., 2018). In this study, the V/Ni ratios in PM_{2.5} ranged from 0.7 to 2.8, showing the mixing of
385 various sources (probably from fossil fuel combustion). A specific trend (V/Ni = 0.15, $r^2 = 0.98$, $n = 3$)
386 was also observed in the spring under easterly winds with V/Ni highlighting the impact of road traffic
387 in the samples (Fig. 5a).

388 Coal combustion, industrial smelting and waste incineration are important
389 sources of Cd and Pb in atmospheric particles (Pacyna and Pacyna, 2001; Sun et
390 al., 2010; Tang et al., 2013; Sun et al., 2014). Generally, Cd/Pb ratios in
391 urban/industrial cities in Asia have values close to 0.033 in Taiwan (Chen et al.,
392 2015), 0.030 in Taipei (Hsu et al., 2005), 0.025 in Beijing and Changchun and
393 0.049 in Chengdu (Xu et al., 2020). In Europe, same ratios (0.029) have also been
394 observed in the steel industry (Oravisjarvi et al., 2003) and higher values (0.08)
395 have been found in municipal waste incinerators (Font et al., 2015). Conversely,
396 lower Cd/Pb ratios have been found in natural Asian soils (0.017; Hsu et al., 2005)
397 or rural European villages (0.013; Font et al., 2015). In this study, the Cd/Pb ratios in
398 PM_{2.5} showed two trends with values close to 0.032 ($r^2 = 0.96$; $n = 48$) or close to 0.012 ($r^2 = 0.99$;
399 $n = 8$) (Fig. 5b). Although the highest trend appears to be independent of weather conditions, the
400 lowest trend was observed with southeasterly winds and rainy days. These results could indicate a
401 dominant emission of PM_{2.5} from local urban/industrial activities throughout the year and a
402 predominance of natural inputs of PM_{2.5} in specific weather conditions. In addition, high Cd/Pb ratio
403 (0.088) was also observed during a specific event (18/02/2020) probably related to waste
404 incineration emissions.

405 Sb is mainly derived from anthropogenic sources, in particular emissions from brake pad
406 wear (Varrica et al., 2013) as well as, more broadly, from industrial activities and fossil fuel
407 combustion (Pacyna and Pacyna, 2001; Tian et al., 2014). A specific Sb/Cd ratio of 5.0 has been
408 attributed to road brake pad wear (Thorpe and Harrison, 2008) A lower Sb/Cd ratio can also be found

409 in Chinese and Vietnamese coal but the value varies by geographical area. Indeed, a Sb/Cd ratio of
410 1.2 was found in the Liupanshui coal mine (Guizhou, southwest China; [Zhuang et al., 2000](#)) and an
411 average of 0.9 (n = 1,123) was proposed for the whole country ([Dai et al., 2012](#)). However, coal mines
412 in northern Vietnam are rich in organic sulphur and enriched with Sb thus increasing the Sb/Cd ratio
413 ([Li et al., 2023](#)). For example, a Sb/Cd ratio of 3.0 was found in the Coc Sau coal mine ([Chifflet et al.,](#)
414 [2018](#)). In this study, high variability in Sb/Cd ratios was observed in PM_{2.5}, indicating a possible mix
415 of different anthropogenic sources (Fig. 5c). Three trends were observed during the sampling period
416 with Sb/Cd ratios close to 4.3 ($r^2 = 0.99$; n = 32), close to 2.6 ($r^2 = 0.99$; n = 14) or close to 1.3
417 ($r^2 = 0.92$; n = 11). Although the weakest trends are observed by east and northeast winds
418 (Sb/Cd = 1.3) or northeast and southeast winds (Sb/Cd = 2.6), the highest trend (Sb/Cd = 4.3) does
419 not appear to be influenced by wind direction. These results seem to confirm the predominance of
420 regional and local PM_{2.5} emissions from coal activities mixed with local PM_{2.5} emissions from road
421 traffic..

422 Rapid urbanisation and industrialisation in southeast Asia, as well as long-range
423 transboundary transport of atmospheric PM, make it difficult to assess the presence of natural
424 sources in this region. In general, natural particles are emitted by wind erosion of soils in arid regions.
425 MM concentrations from pristine soil in Vietnam are missing. However, soil dust appears to be the
426 dominant source of Al and Ti in Hanoi ([Hein et al., 2022](#)) and a significant Al/Ti ratio of 14.5 ($r^2 = 0.98$;
427 n = 54) is observed in this study (Fig. 5d). Same Al/Ti ratio (14.3) was also found in PM_{2.5} collected
428 during sand dust storms in northern Chinese megacities ([Xu et al., 2004a](#); [Shen et al., 2007](#)) showing
429 a possible long-range transport of atmospheric PM. In Seoul (North Korea), the variation in the Al/Ti
430 ratio was closely related to atmospheric transport of particles from the Gobi Desert through China
431 and Mongolia (Al/Ti = 23.5) or particles emitted locally by soil resuspension (Al/Ti = 15.2) ([Kim et al.,](#)
432 [2003](#)). In addition, an average Al/Ti of 25 was proposed for the upper continental crust ([Wedepohl,](#)
433 [1995](#)). These results show the likely mix between natural dust and urban dust in this study.

434

4.2 Effect of local and regional air mass transport on source distribution

4.2.1 Impact of weather conditions on MM concentrations

PM typically persists in the atmosphere between a few days and several weeks before being scavenged by dry or wet deposition (Seinfeld 2014). During their lifetime, they may travel long distances between the source of emission and the deposit area, undergoing chemical and physical transformations. Therefore, weather conditions can strongly influence their concentrations and the MM. In this study, the PCA-HCA analyses showed that high MM concentrations in cluster C2 occurred with north or northeast winds during the winter monsoon (Fig. 4). In addition, in C2, PM_{2.5} concentrations increased as the wind speed decreased (Fig. S2). Such results have already been observed in Hanoi (Hai and Kim Oanh, 2013; Nguyen et al., 2022; Makkoken et al., 2023). This pattern is attributed to recurring temperature inversion events in air layers in winter monsoon, thus limiting the dispersion of PM (Hien et al., 2002; Ly et al. 2021).

In winter, a ridge of high-pressure flows air masses from Siberia to China towards northern Vietnam, bringing cold and dry air, enriched by anthropogenic particles during their long-range transport. However, this is not the only pathway that controls the transport of particles in Hanoi. The height of the boundary layer at the Earth's surface also plays an important role in vertical mixing. Previous studies have shown that high concentrations of elemental carbon (*i.e.*, black carbon produced by incomplete combustion of fossil fuels) reduces vertical mixing (Wilcox et al., 2016; Hu et al., 2020; Duc et al., 2023). This is because the black carbon emitted in the upper air mass absorbs the sunlight, which heats up faster than the air mass closest to the Earth's surface (Ding et al., 2016). As a result, black carbon induces a reduction in turbulent kinetic energy in the surface mixed layer, which promotes its enrichment by local particle emissions (Petäjä et al., 2016). Conversely, during the summer monsoon, winds from the southeastern South China Sea and the sub-tropical western Pacific carry warm, moist, particle-depleted air. High temperatures enhance strong atmospheric convection, thus improving the dispersion of anthropogenic emissions (Hein et al., 2002; Yang et al.

460 2018). In addition, heavy precipitation during the intermediate fall periods further decreases PM2.5
461 (and bound MM) due to the scavenging effects (Wang et al. 2018).

462

463 4.2.2 Temporal variation of sources

464 The predominant origin of Pb in PM2.5 was further evidenced by the plotting of Pb isotopic
465 ratios ($^{208}\text{Pb}/^{206}\text{Pb}$ vs $^{207}\text{Pb}/^{206}\text{Pb}$) from different anthropogenic sources (urban traffic and unleaded
466 gasoline, urban/industrial activities and coal mining) of neighbouring countries (*i.e.*, China, India,
467 Indonesia, Taiwan, Thailand) (Fig. 6, Table S8). Lead in atmospheric particles has been extensively
468 studied in Asian countries over the last two decades and has shown specific isotopic variations
469 depending on the emission source (Bing-Quan et al., 2001; Bollhöfer and Rosman, 2001; Mukai et al.,
470 2001a,b; Zhu et al., 2001; Zheng et al., 2004; Yao et al., 2015; Zhao et al., 2015; Bi et al., 2017). Here,
471 the mixing between three sources is represented by a triangle whose end-members define the Pb
472 isotopic ratios in sources S1, S2 and S3. The end-member S1, less radiogenic than other
473 anthropogenic samples, could represent the geochemical background derived from the Earth's crust
474 (Bollhöfer and Rosman, 2001; Li et al., 2001). However, due to the high heterogeneity of Asian soils
475 and the intensity of anthropogenic activities, the measurement of the geochemical background is
476 uncertain. It is highly probable that natural dusts from long-range transport are contaminated by
477 deposition of anthropogenic emissions (Table 2). The Pb isotopic composition in S1 was compared to
478 ceramic or metals assemblies from neolithic archaeological site in southeast Asian (Pryce et al., 2014).
479 The relics had Pb isotopic signatures close to S1, thus confirming the latter as a natural background.
480 Therefore, Pb isotopic composition in 'natural' soil (S1) were assessed by calculating the intercept
481 between the "coal line" and the "oil line". The Pb isotopic ratios in S2 were associated with those of
482 the Coc Sau coal mine (northern Vietnam; Chifflet et al., 2018) which merges with other Chinese coal
483 mines (the solid green circles specifically representing the southern mines). Finally, the end-member
484 S3 was associated with values from vehicle exhaust emissions in the megacity of Bangkok (Thailand,
485 Bing-Quan et al., 2001) which also merges with those from Chinese unleaded gasoline (Yao et al.,

486 [2015](#)) and Russian oil fields ([Mukai et al., 2001b](#)). The orange linear trend is defined with Pb isotopic
487 ratios measured in samples from anthropogenic petroleum-related activities labelled S3 'oil', here
488 after.

489 The contributions of the three Pb sources in PM_{2.5} was calculated using a linear mixing
490 model (Fig. 7, Table S9). A low contribution of 'natural' soil ($14 \pm 11\%$) and high PM_{2.5} concentrations
491 ($54 \pm 30 \mu\text{g m}^{-3}$) were observed during the northeast winter monsoon. Conversely, the contribution
492 of 'natural' soil increased sharply (up to 83% on 07/07/2020) with a decrease in PM_{2.5}
493 concentrations ($26 \pm 11 \mu\text{g m}^{-3}$) during the southeast summer monsoon. In winter, this contribution
494 can be attributed to Asian dust storms, which transport fine soil dust from eastern (or inland) China
495 to northern Vietnam before reaching Hanoi. Locally, the dry climate favours the resuspension of
496 street dust and fine particles from vehicle exhaust ([Hein et al., 2002](#)). In addition, during this period,
497 air masses with low vertical mixing favour PM_{2.5} enrichment ([Ngoc et al., 2021](#)), thus decreasing the
498 natural contribution of PM_{2.5} from long-range transport. Conversely, during the summer monsoon,
499 the increase in natural contribution of PM_{2.5} can be attributed to both mixing with a cleaner marine
500 airflow and scavenging of local anthropogenic emissions due to a wet climate. The 'oil' contribution
501 (encompassing any use) in PM_{2.5} remained stable over the sampling period ($48 \pm 13\%$ during the
502 northeast winter monsoon and $42 \pm 17\%$ during the southeast summer monsoon), although the year
503 2020 is marked by the traffic restrictions imposed by COVID-19 and a possible reduction in fuel
504 consumption in many motorists (*e.g.*, vehicles, buses, trucks *etc.*). In China, almost one third of the
505 cities were in lockdown for several weeks in January and February 2020, resulting in improved local
506 air quality ([He et al., 2020](#); [Yuan et al., 2021](#)). Therefore, during this period (winter monsoon), Hanoi
507 may have been less impacted by transboundary oil emissions. In addition, during the summer
508 monsoon, South China Sea winds are expected to decrease the anthropogenic contribution of
509 atmospheric particles in Hanoi (*via* dilution and wet scavenging) which does not seem to be the case
510 for the 'oil' contribution which remains stable over the year. These observations suggest that 'oil'
511 contribution in PM_{2.5} was mainly of local origin, not influenced by long-range transport from China

512 or South China Sea. In contrast, coal had a higher contribution in PM_{2.5} during the northeast winter
513 monsoon ($38 \pm 13\%$) than during southeast summer monsoon ($26 \pm 15\%$). Despite the rapid
514 development of green energy (wind and solar), CO₂ emissions continue to rise in China. In 2020, the
515 Chinese coal industry emitted 761 Mt CO₂ (*i.e.*, 70% of China's total CO₂ emissions) due to the
516 growing demand for electricity (IEA, 2020; <https://ourworldindata.org/>). In addition, Vietnam has
517 also large coal reserves in the north of the country and the share of coal-fired power generation has
518 increased to 50% by 2020 (<https://ember-climate.org/>). Therefore, the results show a latent local
519 coal contribution in PM_{2.5} (about 26%) enriched by Chinese coal emissions (about +12%) in winter.

520

521 *4.3 Human health risk assessment through exposure to PM_{2.5}*

522 In this study, PM_{2.5} contain high concentrations of MM, which may constitute a health risk.
523 Therefore, a comprehensive human health risk assessment can provide useful information for public
524 policy management. The hazard quotient (HQ_i) per element could be ranked in the following order:
525 Cr > Pb > Sb > Zn > Cu > Mn > As > Ti > Cd > V > Ni > Al > Co > Hg > U. The cumulative hazard index
526 (HI) was less than 1 (*i.e.*, 0.076), suggesting that inhalation of the 15 elements bound to PM_{2.5} has no
527 chronic influence (Table 3). In addition, the total carcinogenic risk (TCR) was estimated at $1.0 \cdot 10^{-4}$,
528 which suggests that the probability of a person developing cancer by inhalation is low, but would be
529 mainly imputable to Cr.

530 Other studies on human health risk assessment of PM_{2.5} in Hanoi estimated risk values
531 similar to our results: HI = 0.13 ± 0.08 and TCR = $1.13 \cdot 10^{-5}$ (Hein et al., 2022), HI = 1.16 and TCR = $9 \cdot 10^{-6}$
532 (Nguyen et al., 2022) or HI = 0.006 and TCR = $5.5 \cdot 10^{-7}$ (Bui et al., 2022). The differences between HI
533 and TCR values may be explained by the random or incomplete sampling frequency, which were
534 carried out only in winter 2018-2019, in three seasons winter-spring-fall 2020 or in two seasons
535 summer-fall 2020, respectively. However, these studies concluded that Cr and As presented a risk of
536 inhalation exposure. According to the International Agency for Research on Cancer, Cr (Group 1) can
537 induce harmful human health effects. Coal naturally contains large amounts of Cr (Zhang et al., 2004;

538 [Ketris and Yudovich, 2009; Dai et al., 2012](#)) which are transferred to atmospheric particles during
539 combustion processes ([Xu et al., 2004b](#)). Consequently, coal mining and combustion could potentially
540 induce human diseases.

541 Despite risks on human health, Vietnam has significantly increased its use of coal, positioned
542 as the world's tenth largest producer of coal-fired electricity. In 2022, Vietnamese economy grew by
543 8% with a very large increase in energy use (<https://ember-climate.org/>). Despite the boom in the
544 deployment of renewables energies, Vietnam faces several challenges (industrial, economic and
545 politic) in its effort to become carbon neutral. A "Just Energy Transition Partnership" plan was signed
546 in 2022 in order to help the country to peak its greenhouse gas emissions by 2030 instead of 2035,
547 limiting its production of coal energy to 30.2 GW instead of 37 GW, and source 47% of its power from
548 renewable energy by 2030 (COP27 climate summit; <https://unfccc.int/cop27>). This study highlights
549 the need for strict regulatory plans to control hazardous MM emissions by reducing coal combustion.
550 In addition, due to the impact of oil combustion in PM2.5, public policies should encourage
551 conversion to green transport to limit global warming.

552

553

554 **5. Conclusion**

555 Our study provided a detailed overview of the annual variation in PM2.5 concentrations in
556 Hanoi (northern Vietnam), evaluated Pb anthropogenic sources and examined the impact of weather
557 conditions on inhaled MM concentrations and related toxic risks.

558 The annual average PM2.5 concentration ($40.2 \pm 26.3 \mu\text{g m}^{-3}$) was 1.6 and 8.0 times higher
559 than the national and global standards, respectively. In addition, some PM2.5-bound MM (Pb, As, Cd
560 and Cr) had 24-hour maximum concentrations above the annual exposure limits. Chemical ratios
561 (V/Ni, Cd/Pb, Sb/Cd and Al/Ti) presented typical trends indicating the dominance of three emission
562 sources from road traffic, natural soil, and coal uses (mining, combustion). The source contribution
563 was assessed using Pb isotopic ratios ($^{208}\text{Pb}/^{206}\text{Pb}$ vs $^{207}\text{Pb}/^{206}\text{Pb}$). The oil contribution (encompassing

564 any petroleum-related activities) to PM2.5 remained stable over the sampling period (November
565 2019 to December 2020) with an annual average of 45%. In contrast, PM2.5 inputs from soil and coal
566 were impacted by weather conditions. Indeed, 'natural' soil contribution varied between 14 to 83%
567 depending on the winter and summer monsoons, respectively.. Conversely, coal contribution was
568 higher in the winter (38%)than in the summer (26%) monsoon. This phenomenon could be due to the
569 BC emissions (from the incomplete combustion of fossil fuels) coming from both long-range air
570 transport and local emissions. In addition, the human health risk assessment showed that PM2.5-
571 bound MM concentrations obtained in this study have no chronic influence although Cr, Pb and Sb
572 have the highest HQ_i values. The carcinogenic risk is low but mainly imputable to Cr.

573 In order to discriminate more precisely the black carbon fractions linked to agricultural
574 burning from those linked to the combustion of fossil fuels, a companion paper will cover the study
575 of major inorganic ions (NH₄⁺, NO₃⁻, SO₄²⁻, Na⁺) and other elements (Ca, K, Mg, P), as well as carbon
576 speciation (OC, EC, sugars).

577

578

579

580 **Supplementary information**

581 The supplementary material related to this article is available online at XXX.

582

583

584 **Acknowledgments**

585 The project received financial support from the French (IRD, IRN-SOOT SEA) and Vietnamese
586 (VAST-IMER) research organizations. We thank the MIO "Plateforme Analytique de Chimie des
587 Environnements Marins" (PACEM platform) for laboratory facilities. The authors would like to thank
588 the dedicated efforts of people from Air-o-Sol Platform at IGE for analysing the PM2.5 samples. The

589 authors are deeply grateful for the thorough and constructive corrections and insightful comments of
590 the anonymous reviewers, who have significantly improved the original manuscript.

591

592

593 **References**

594 Bacon, S.N., McDonald, E.V., Amit, R., Enzel, Y., Crouvi, O., 2011. Total suspended particulate matter
595 emissions at high friction velocities from desert landforms. *J. Geophys. Res.*, 116, F03019.
596 <https://doi.org/10.1029/2011JF001965>.

597 Balali-Mood, M., Naseri, K., Tahergorabi, Z., Khazdair, M.R., Sadeghi, M., 2021 Toxic mechanisms of
598 five heavy metals: mercury, lead, chromium, cadmium, and arsenic. *Front. Pharmacol.*, 12,
599 643972. <https://doi.org/10.3389/fphar.2021.643972>.

600 Behrooz, R.D., Kaskaoutis, D.G., grivas, G., Mihalopoulos, N., 2021. Human health risk assessment for
601 toxic elements in the extreme ambient dust conditions observed in Sistan, Iran. *Chemosphere*,
602 262, 127835. <https://doi.org/10.1016/j.chemosphere.2020.127835>.

603 Bi, X.Y., Li, Z.G., Wang, S.X., Zhang, L., Xu, R., Liu, J.L., Yang, H.M., Guo, M.Z., 2017. Lead Isotopic
604 Compositions of Selected Coals, Pb/Zn Ores and Fuels in China and the Application for Source
605 Tracing. *Environ. Sci. Technol.*, 51, 13502-13508. <https://doi.org/10.1021/acs.est.7b04119>.

606 Bing-Quan, Z., Yu-Wei, C., Jian-Hua, P., 2001. Lead isotope geochemistry of the urban environment in
607 the Pearl River Delta. *App. Geochem.*, 16, 409-417. [https://doi.org/10.1016/S0883-
608 2927\(00\)00047-0](https://doi.org/10.1016/S0883-2927(00)00047-0).

609 Bing-Quan, Z., Yu-Wei, C., Xiang-yang, C., 2002. Application of Pb isotopic mapping to environment
610 evaluation in China. *Chem. Speciat. Bioavail.*, 14, 49-56.
611 <https://doi.org/10.3184/095422902782775335>.

612 Bollhöfer, A., Rosman, K.J.R., 2000. Isotopic source signatures for atmospheric lead: the Southern
613 hemisphere. *Geochim. Cosmochim. Acta*, 64, 3251-3262. [https://doi.org/10.1016/S0016-
614 7037\(00\)00436-1](https://doi.org/10.1016/S0016-7037(00)00436-1).

615 Bollhöfer, A, Rosman, K.J.R., 2001. Isotopic source signatures for atmospheric lead: the Northern
616 Hemisphere. *Geochim. Cosmochim. Acta*, 65, 1727-1740. [https://doi.org/10.1016-](https://doi.org/10.1016/S0016-7037(00)00630-X)
617 [7037\(00\)00630-X](https://doi.org/10.1016/S0016-7037(00)00630-X).

618 Bui, T.H., Nguyen, D.L., Nguyen, T.P.M., Bui, Q.T., Nguyen, V.D., Mac, V.D., 2022. Chemical
619 characterization, source apportionment, and human health risk assessment of PM_{2.5} in an
620 urban area in Hanoi, Vietnam. *Air Qual. Atmos. Health*, 16, 149-163.
621 <https://doi.org/10.1007/s11869-022-01266-0>.

622 Chen, M., Ma, L.Q., 2001. Comparison of Three Aqua Regia Digestion Methods for Twenty Florida
623 Soils. *Soil Sci. Soc. Am. J.*, 65, 491-499. <https://doi.org/10.2136/sssaj2001.652491x>.

624 Chen, Y.C., Hsu, C.Y., Lin, S.L., Chang-Chien, G.P., Chen, M.J., Fang, G.C., Chiang, H.C., 2015.
625 Characteristics of Concentrations and Metal Compositions for PM_{2.5} and PM_{2.5-10} in Yunlin
626 County, Taiwan during Air Quality Deterioration. *Aerosol Air Qual. Res.*, 15, 2571-2583.
627 <https://doi.org/10.4209/aaqr.2015.04.0261>.

628 Cheng, I., Al Mamun, A., Zhang, L., 2021. A synthesis review on atmospheric wet deposition of
629 particulate elements: scavenging ratios, solubility, and flux measurements. *Environ. Rev.*, 29,
630 340-353. <https://doi.org/10.1139/er-2020-0118>.

631 Chifflet, S., Amouroux, D., Bérail, S., Barre, J., Thuoc, T.V., Baltrons, O., Brune, J., Dufour, A., Guinot,
632 B., Mari, X., 2018. Origins and discrimination between local and regional atmospheric pollution
633 in Haiphong (Vietnam), based on metal(loid) concentrations and lead isotopic ratios in PM₁₀.
634 *Environ Sci Pollut Res*, 25, 26653-26668. <https://doi.org/10.1007/s11356-018-2722-7>.

635 Chow, J.C., Lowenthal, D.H., Chen, L.W.A., Wang, X., Watson, J.G., 2015. Mass reconstruction
636 methods for PM_{2.5}: a review. *Air Qual. Atmos. Health*, 8, 243-263.
637 <https://doi.org/10.1007/s11869-015-0338-3>.

638 Cohen, D.D., Crawford, J., Stelcer, E., Bac, T.V., 2010. Characterisation and source apportionment of
639 fine particulate sources at Hanoi from 2001 to 2008. *Atmos. Environ.*, 44, 320-328.
640 <https://doi.org/10.1016/j.atmosenv.2009.10.037>.

641 Dai, S., Ren, D., Chou, C.L., Finkelman, R.B., Seredin, V.V., Zhou, Y., 2012. Geochemistry of trace
642 elements in Chinese coals: A review of abundances, genetic types, impacts on human health,
643 and industrial utilization. *Int. J. Coal Geol.*, 94, 3-21.
644 <https://doi.org/10.1080/00206814.2017.1405287>.

645 Ding, A.J., Huang, X., Nie, W., Sun, J.N., Kerminen, V.-M., Petäjä, T., Su, H., Cheng, Y.F., Yang, X.-Q.,
646 Wang, M.H., Chi, X.G., Wang, J.P., Virkkula, A., Guo, W.D., Yuan, J., Wang, S.Y., Zhang, R.J., Wu,
647 Y.F., Song, Y., Zhu, T., Zilitinkevich, S., Kulmala, M., Fu, C.B., 2016. Enhanced haze pollution by
648 black carbon in megacities in China. *Geophys. Res. Lett.*, 43.
649 <https://doi.org/10.1002/2016GL067745>.

650 Draxler, R.R., Stunder, B., Rolph, G., Stein, A., Taylor, A., Zinn, S., Loughner, C., Crawford, A., 2022.
651 HYSPLIT User's Guide. Version 5.2, Last revision: Jan 2022.
652 https://www.arl.noaa.gov/documents/reports/hysplit_user_guide.pdf.

653 Duan, X., Yan, Y., Li, R., Deng, M., Hu, D., Peng, L., 2021. Seasonal variations, source apportionment,
654 and health risk assessment of trace metals in PM_{2.5} in the typical industrial city of Changzhi,
655 China. *Atmos. Pollut. Res.*, 12, 365-374. <https://doi.org/10.1016/j.apr.2020.09.017>.

656 Duc, H.N., Azzi, M., Zhang, Y., Kirkwood, J., White, S., Trieu, T., Riley, M., Salter, D., Chang, L.T.C.,
657 Capnerhurst, J., Ho, J., Gunashanhar, G., Monk, K., 2023. Black Carbon Emissions, Transport
658 and Effect on Radiation Forcing Modelling during the Summer 2019-2020 Wildfires in
659 Southeast Australia. *Atmosphere*, 14, 699. <https://doi.org/10.3390/atmos14040699>.

660 Finlayson-Pitts, B.J., Wingen, L.M., Perraud, V., Ezzll, M.J., 2020. Open questions on the chemical
661 composition of airborne particles. *Commun. Chem.*, 3, 108. [https://doi.org/10.1038/s42004-](https://doi.org/10.1038/s42004-020-00347-4)
662 [020-00347-4](https://doi.org/10.1038/s42004-020-00347-4).

663 Font, A., de Hoogh, K., Leal-Sanchez, M., Ashworth, D.C., Brown, R.J.C., Hansell, A.L., Fuller, G.W.,
664 2015. Using metal ratios to detect emissions from municipal waste incinerators in ambient air
665 pollution data. *Atmos. Environ.*, 113, 177-186.
666 <http://dx.doi.org/10.1016/j.atmosenv.2015.05.002>.

667 Fuzzi, S., Baltensperger, U., Carslaw, K., Decesari, S., Denier van der Gon, H., Facchini, M. C., Fowler,
668 D., Koren, I., Langford, B., Lohmann, U., Nemitz, E., Pandis, S., Riipinen, I., Rudich, Y., Schaap,
669 M., Slowik, J. G., Spracklen, D. V., Vignati, E., Wild, M., Williams, M., and Gilardoni, S., 2015.
670 Particulate matter, air quality and climate: lessons learned and future needs. *Atmos. Chem.*
671 *Phys.*, 15, 8217-8299. <https://doi.org/10.5194/acp-15-8217-2015>.

672 Gieré, R., Querol, X., 2010. Solid Particulate Matter in the Atmosphere. *Elements*, 6, 215-222.
673 <https://doi.org/10.2113/gselements.6.4.215>.

674 Gilbert, R.O., 1987. *Statistical Methods for Environmental Pollution Monitoring*. John Wiley & Sons,
675 Inc., New York, pp. 177-185. ISBN: 978-0-471-28878-7.

676 Guo, F., Tang, M., Wang, X., Yu, Z., Wei, F., Zhang, X., Jin, M., Wang, J., Xu, D., Chen, Z., Chen, K., 2022.
677 Characteristics, sources, and health risks of trace metals in PM_{2.5}. *Atmos. Environ.*, 289,
678 119314. <https://doi.org/10.1016/j.atmosenv.2022.119314>.

679 Guo, S., Tao, X., Liang, L., 2023. Exploring Natural and Anthropogenic Drivers of PM_{2.5} Concentrations
680 Based on Random Forest Model: Beijing–Tianjin–Hebei Urban Agglomeration, China.
681 *Atmosphere*, 14, 381. <https://doi.org/10.3390/atmos14020381>.

682 Hai, C.D., Kim Oanh, N.T., 2013. Effects of local, regional meteorology and emission sources on mass
683 and compositions of particulate matter in Hanoi. *Atmos. Environ.* 78, 105-112.
684 <https://doi.org/10.1016/j.atmosenv.2012.05.006>.

685 Hao, Y., Meng, X., Yu, X., Lei, M., Li, W., Shi, F., Yang, W., Zhang, S., Xie, S., 2018. Characteristics of
686 trace elements in PM_{2.5} and PM₁₀ of Chifeng, northeast China: Insights into spatiotemporal
687 variations and sources. *Atmos. Res.*, 213, 550-561.
688 <https://doi.org/10.1016/j.atmosres.2018.07.006>.

689 He, G., Pan, Y., Tanaka, T., 2020. The short-term impacts of COVID-19 lockdown on urban air
690 pollution in China. *Nat. Sustain.*, 3, 1005–1011. <https://doi.org/10.1038/s41893-020-0581-y>.

691 Henry R.C., Lewis C.W., Hopke P.K., Williamson H.J., 1984. Review of receptor model fundamentals.
692 *Atmos. Environ.*, 18, 1507-1515. [https://doi.org/10.1016/0004-6981\(84\)90375-5](https://doi.org/10.1016/0004-6981(84)90375-5).

693 Hien, P.D., Bac, V.T., Tham, H.C., Nhan, D.D., Vinh, L.D., 2002. Influence of meteorological conditions
694 on PM_{2.5} and PM₁₀ concentrations during the monsoon season in Hanoi, Vietnam. *Atmos.*
695 *Environ.*, 36, 3473–3484. [https://doi.org/10.1016/S1352-2310\(02\)00295-9](https://doi.org/10.1016/S1352-2310(02)00295-9).

696 Hien, P.D., Men, N.T., Tan, P.M., Hangartner, M., 2020. Impact of urban expansion on the air
697 pollution landscape: a case study of Hanoi, Vietnam. *Sci. Total Environ.*, 702, 134635.
698 <https://doi.org/10.1016/j.scitotenv.2019.134635>.

699 Hien, T.T., Chi N.C.T., Huy, D.H., Le, H.A., Oram, D.E., Forster, G.L., Mills, G.P., Baker, A.R., 2022.
700 Soluble trace metals associated with atmospheric fine particulate matter in the two most
701 populous cities in Vietnam. *Atmos. Environ.*, 15, 100178.
702 <https://doi.org/10.1016/j.aeaoa.2022.100178>.

703 Hopke, P. K., 2000. A guide to positive matrix factorization. In: Workshop on UNMIX and PMF as
704 applied to PM_{2.5}, pp 600.

705 Hopke, P.K., Cohen, D.D., Begum, B.A., Biswas, S.K., Ni, B., Pandit, G.G., Santoso, M., Chung, Y.-S.,
706 Davy, P., Markwitz, A., Waheed, S., Siddique, N., Santos, F.L., Pabroa, P.C.B., Seneviratne,
707 M.C.S., Wimolwattanapun, W., Bunrapob, S., Vuong, T.B., Duy Hien, P., Markowicz, A., 2008.
708 Urban air quality in the Asian region. *Sci. Total Environ.* 404, 103–112.
709 <https://doi.org/10.1016/j.scitotenv.2008.05.039>

710 Hopke, P.K., Cohen, D.D., 2011. Application of receptor modeling methods. *Atmos. Pollut. Res.*, 2,
711 122-125. <https://doi.org/10.5094/APR.2011.016>.

712 Hsu, S.C., Liu, S.C., Jeng, W.L., Lin, F.J., Huang, Y.T., Lung, S.C.C., Liu, T.H., Tu, J.Y., 2005. Variations of
713 Cd/Pb and Zn/Pb ratios in Taipei aerosols reflecting long-range transport or local pollution
714 emissions. *Sci. Tot. Environ.*, 347, 111-121. <https://doi.org/10.1016/j.scitotenv.2004.12.021>.

715 Hu, X., Zhang, Y., Ding, Z., Wang, T., Lian, H., Sun, Y., Wu, J., 2012. Bioaccessibility and health risk of
716 arsenic and heavy metals (Cd, Co, Cr, Cu, Ni, Pb, Zn and Mn) in TSP and PM_{2.5} in Nanjing,
717 China. *Atmos. Environ.*, 57, 146–152. <https://doi.org/10.1016/j.atmosenv.2012.04.056>.

718 Hu, K., Zha, D., Liu, D., Ding, S., Tian, P., Yu, C., Zhou, W., Huang, M., Ding, D., 2020. Estimating
719 radiative impacts of black carbon associated with mixing state in the lower atmosphere over
720 the northern North China Plain. *Chemosphere*, 252, 126455.
721 <https://doi.org/10.1016/j.chemosphere.2020.126455>.

722 Huu, D.N., Ngoc, V.N., 2021. Analysis study of current transportation status in Vietnam's urban traffic
723 and the transition to electric two-wheelers mobility. *Sustainability*, 13, 5577.

724 Huy, L.N., Kim Oanh, N.T., 2017. Assessment of national emissions of air pollutants and climate
725 forcers from thermal power plants and industrial activities in Vietnam. *Atmos. Pollut. Res.*, 8,
726 503-513. <https://doi.org/10.1016/j.apr.2016.12.007>.

727 IEA (International Energy Agency), 2020. *Energy Technology Perspectives 2020*, IEA, Paris 400 p.

728 IARC, 2023. *Monographs on the Evaluation of Carcinogenic Risks to Humans Volume 1-133*.
729 <https://monographs.iarc.who.int/agents-classified-by-the-iarc/>.

730 Jaishankar, M., Tseten, T., Anbalagan, N., Mathew, B.B., Beeregowda, K.N., 2014. Toxicity,
731 mechanism and health effects of some heavy metals. *Interdiscip. Toxicol.*, 7, 60-72.
732 <https://doi.org/10.2478/intox-2014-0009>.

733 Jaffrezo, J.L., Colin, J.L., 1988. Rain-aerosol coupling in urban area: Scavenging ratio measurement
734 and identification of some transfer processes. *Atmos. Environ.*, 22, 929-935.
735 [https://doi.org/10.1016/0004-6981\(88\)90270-3](https://doi.org/10.1016/0004-6981(88)90270-3).

736 Janssen, N.A.H., Hoek, G., Simic-Lawson, M., Fischer, P., van Bree, L., ten Brink, H., Keuken, M.,
737 Atkinson, R.W., Anderson, H.R., Brunekreef, B., Cassee, F.R., 2011. Black Carbon as an
738 Additional Indicator of the Adverse Health Effects of Airborne Particles Compared with PM₁₀
739 and PM_{2.5}. *Environmental Health Perspectives*, 119, 12. <https://doi.org/10.1289/ehp.1003369>.

740 Ketris, M.P.; Yudovich, Ya.E., 2009. Estimation of clarkes for carbonaceous biotithes: World averages
741 for trace element contents in black shales and coals. *Int. J. Coal Geol.*, 78, 135-148.
742 <https://doi.org/10.1016/j.coal.2009.01.002>.

743 Kikuchi, T., Hai, H.T., Tanaka, S., 2010. Characterization of heavy metal contamination from their
744 spatial distribution in sediment of an urban lake of Hanoi, Vietnam. *J. Water Environ. Technol.*,
745 8, 111-123. <https://doi.org/10.2965/jwet.2010.111>.

746 Kim, K.H., Choi, G.H., Kang, C.H., Lee, J.H., Kim, J.Y., Youn, Y.H., Lee, S.R., 2003. The chemical
747 composition of fine and coarse particles in relation with the Asian Dust events. *Atmos.*
748 *Environ.*, 37, 753-765. [https://doi.org/10.1016/S1352-2310\(02\)00954-8](https://doi.org/10.1016/S1352-2310(02)00954-8).

749 Komárek, M., Ettler, V., Chrastny, V., Milhaljevic, M., 2008. Lead isotopes in environmental sciences:
750 a review. *Environ. Int.*, 34, 562-577. <https://doi.org/10.1016/j.envint.2007.10.005>.

751 Laskin, A., Moffet, R.C., Gilles, M.K., 2019. Chemical imaging of atmospheric particles. *Acc. Chem.*
752 *Res.*, 52, 3419-3431. <https://doi.org/10.1021/acs.accounts.9b00396>.

753 Lasko, K., Vadrevu, K., 2018. Improved rice residue burning emissions estimates: Accounting for
754 practice-specific emission factors in air pollution assessments of Vietnam. *Environ. Pollut.*, 236,
755 795-806. <https://doi.org/10.1016/j.envpol.2018.01.098>.

756 Lelieveld, J., Pozzer, A., Pöschl, U., Fnais, M., Haines, A., Münzel, T., 2020. Loss of life expectancy from
757 air pollution compared to other risk factors: a worldwide perspective. *Cardiovascular research*,
758 116, 1910-1917. <https://doi.org/10.1093/cvr/cvaa025>.

759 Le, H.A., Phuong, D.M., Linh, L.T., 2020. Emission inventories of rice straw open burning in the Red
760 River Delta of Vietnam: evaluation of the potential of satellite data. *Environ. Pollut.*, 260,
761 113972 <https://doi.org/10.1016/j.envpol.2020.113972>.

762 Lee, S.W., Pomalis, R., Kan, B., 2000. A new methodology for source characterization of oil
763 combustion particulate matter. *Fuel Process. Technol.*, 65-66, 189-202.
764 [https://doi.org/10.1016/S0378-3820\(99\)00086-7](https://doi.org/10.1016/S0378-3820(99)00086-7).

765 Li, J., Okin, G.S., Epstein, H.E., 2009. Effects of enhanced wind erosion on surface soil texture and
766 characteristics of windblown sediments. *J. Geophys. Res.*, 114, G02003.
767 <https://doi.org/10.1029/2008JG000903>.

768 Li, F., Yan, J., Wei, Y., Zeng, J., Wang, X., Chen, X., Zhang, C., Li, W., Chen, M., Lü, G., 2021. PM2.5-
769 bound heavy metals from the major cities in China: Spatiotemporal distribution, fuzzy
770 exposure assessment and health risk management. *J. Clean. Prod.*, 286, 124967.
771 <https://doi.org/10.1016/j.jclepro.2020.124967>.

772 Li, J., Lin, Y., Nguyen, X., Zhuang, X., Li, B., Querol, X., Moreno, N., Cordoba, P., 2023. Enrichment of
773 strategic metals in the Upper Triassic coal from the Nui Hong open-pit mine, Thai Nguyen
774 Coalfield, NE Vietnam. *Ore Geol. Rev.*, 153, 105301.
775 <https://doi.org/10.1016/j.oregeorev.2023.105301>.

776 Liao, K., Zhang, J., Chen, Y., Lu, X., Fung, J.C.H., Ying, Q., Yu, J.Z., 2023. Regional source apportionment
777 of trace metals in fine particulate matter using an observation-constrained hybrid model. *Clim.*
778 *Atmos. Sci.*, 6, 65. <https://doi.org/10.1038/s41612-023-00393-4>.

779 Ly, B.T., Matsumi, Y., Vu, T.V., Sekiguchi, K., Nguyen, T.T., Pham, C.T., Nghiem, T.D., Ngo, I.H.,
780 Kurotsuchi, Y., Nguyen, T.H., Nakayama, T., 2021. The effects of meteorological conditions and
781 long-range transport on PM2.5 levels in Hanoi revealed from multi-site measurement using
782 compact sensors and machine learning approach. *J. Aerosol Sci.*, 152, 105716.
783 <https://doi.org/10.1016/j.jaerosci.2020.105716>.

784 Makkonen, U., Vestenius, M., Huy, L.N., Anh, N.T.N., Linh, P.T.V., Thuy, P.T., Phuong, H.T.M., Nguyen,
785 H., Thuy, L.T., Aurela, M., Hellén, H., Loven, K., Kouznetsov, R., Kyllönen, K., Teinilä, K., Kim
786 Oanh, N.T., 2023. Chemical composition and potential sources of PM2.5 in Hanoi. *Atmos.*
787 *Environ.*, 299, 119650. <https://doi.org/10.1016/j.atmosenv.2023.119650>.

788 Mazzei, F., D'Alessandro, A., Lucarelli, F., Nava, S., Prati, P., Valli, G., Vecchi, R., 2008. Characterization
789 of particulate matter sources in an urban environment. *Sci. Tot. Environ.*, 401, 81-89.
790 <https://doi.org/10.1016/j.scitotenv.2008.03.008>.

791 McNeill, J., Snider, G., Weagle, C.L., Walsh, B., Bissonnette, P., Stone, E., Abboud, I., Akoshile, C., Anh,
792 N.X., Balasubramanian, R., Brook, J.R., Coburn, C., Cohen, A., Dong, J., Gagnon, G., Garland,
793 R.M., He, K., Holben, B.N., Kahn, R., Kim, J.S., Lagrosas, N., Lestari, P., Liu, Y., Jeba, F., Joy, K.J.,

794 Martins, J.V., Misra, A., Norford, L.K., Quel, E.J., Salam, A., Schichtel, B., Tripathi, S. N., Wang,
795 C., Zhang, Q., Brauer, M., Gibson, M.D., Rudich, Y., Martin, R.V., 2020. Large global variations in
796 measured airborne metal concentrations driven by anthropogenic sources. *Sci Rep.*, 10, 21817.
797 <https://doi.org/10.1038/s41598-020-78789-y>.

798 Mitra, S., Chakraborty, A.J., Tareq, A.M., Emran, T.B., Nainu, F., Khusro, A., Idris, A.M., Khandaker,
799 M.U., Osman, H., Alhumaydhi, F.A., Simal-Gandara, J., 2022. Impact of heavy metals on the
800 environment and human health: Novel therapeutic insights to counter the toxicity. *J. King Saud*
801 *Univ. Sci.*, 34, 101865. <https://doi.org/10.1016/j.jksus.2022.101865>

802 Monna, F., Lancelot, J., Croudace, I.W., Cundy, A.B., Lewis, J.T., 1997. Pb isotopic composition of
803 airborne particulate material from France and the Southern United Kingdom: implications for
804 Pb pollution sources in urban areas. *Environ. Sci. Technol.*, 31, 2277-2286.
805 <https://doi.org/10.1021/es960870+>.

806 Mukai, H., Tanaka, A., Fujii, T., Zeng, Y., Hong, Y., Tang, J., Guo, S., Xue, H., Sun, Z., Zhou, J., Xue, D.,
807 Zhao, J., Zhai, G., Gu, J., Zhai, P., 2001a. Regional characteristics of sulphur and lead isotope
808 ratios in the atmosphere at several Chinese urban sites. *Environ. Sci. Technol.*, 35, 1064-1071.
809 <https://doi.org/10.1021/es001399u>.

810 Mukai, H., Machida, T., Tanaka, A., Vera, Y.P., Uematsu, M., 2001b. Lead isotope ratios in the urban
811 air of eastern and central Russia. *Atmos. Environ.*, 35, 2783-2793.
812 [https://doi.org/10.1016/S1352-2310\(00\)00341-1](https://doi.org/10.1016/S1352-2310(00)00341-1).

813 Ngoc, B.A.P., Delbarre, H., Deboudt, K., Dieudonné, E., Tran, D.N., Thanh, S.L., Pelon, J., Ravetta, F.,
814 2021. Key factors explaining severe air pollution episodes in Hanoi during 2019 winter season.
815 *Atmos. Pollut. Res.*, 12, 101068. <https://doi.org/10.1016/j.apr.2021.101068>.

816 Ngu, N.D., Hieu, N.T., 2004. Climate and Climate Resources of Vietnam (Khi Hau Va Tai Nguyen Khi
817 Hau, Vietnam). IHA. Agriculture Publisher, Hanoi, Vietnam.

818 Nguyen, T.H., Nagashima, T., Doan, Q.V., 2020. Air Quality Modelling Study on the Controlling Factors
819 of Fine Particulate Matter (PM_{2.5}) in Hanoi: A Case Study in December 2010. *Atmosphere*, 11,
820 733. <https://doi.org/10.3390/atmos11070733>.

821 Nguyen, T.P.M., Bui, T.H., Nguyen, M.K., Ta, T.N., Tran, T.M.H., Nguyen, Y.N., Nguyen, T.H., 2022.
822 Assessing pollution characteristics and human health risk of exposure to PM_{2.5}-bound trace
823 metals in a suburban area in Hanoi, Vietnam. *Human and Ecological Risk Assessment: An*
824 *International Journal*, 28, 433-454. <https://doi.org/10.1080/10807039.2022.2056872>.

825 OEHHA, 2019. California department of toxic substances control (DTSC). Human and ecological risk
826 office (HERO). Human health risk assessment (HHRA). HERO HHRA Note n°10, 25 February
827 2019. [https://dtsc.ca.gov/wp-content/uploads/sites/31/2019/02/HHRA-Note-10-2019-02-](https://dtsc.ca.gov/wp-content/uploads/sites/31/2019/02/HHRA-Note-10-2019-02-25.pdf)
828 [25.pdf](https://dtsc.ca.gov/wp-content/uploads/sites/31/2019/02/HHRA-Note-10-2019-02-25.pdf)

829 Okuda, T., Kato, J., Mori, J., Tenmoku, M., Suda, Y., Tanaka, S., He, K., Ma, Y., Yang, F., Yu, X., Duan, F.,
830 Lei, Y., 2004. Daily concentrations of trace metals in aerosols in Beijing, China, determined by
831 using inductively coupled plasma mass spectrometry equipped with laser ablation analysis, and
832 source identification of aerosols. *Sci. Tot. Environ.*, 330, 145-158.
833 <https://doi.org/10.1016/j.scitotenv.2004.04.010>.

834 Oravisjarvi, K., Timonen, K.L., Wiikinkoski, T., Ruuskanen, A.R., Heinanen, K., Ruuskanen, J., 2003.
835 Source contributions to PM_{2.5} particles in the urban air of a town situated close to a steel
836 works. *Atmos. Environ.*, 37, 1013-1022. [https://doi.org/10.1016/S1352-2310\(02\)01048-8](https://doi.org/10.1016/S1352-2310(02)01048-8).

837 Ortega, G.S., Pécheyran, C., Bérail, S., Donard, O.F., 2012. A fit-for purpose procedure for lead
838 isotopic ratio determination in crude oil, asphaltene and kerogen samples by MC-ICP-MS. *J.*
839 *Anal. At. Spectrom.*, 27, 1447-1456. <https://doi.org/10.1039/C2JA30143A>.

840 Paatero, P., Tapper, U., 1994. Positive Matrix Factorisation: a non-negative factor model with optimal
841 utilisation of error estimates of data values. *Environmetrics*, 5, 111-126.
842 <https://doi.org/10.1002/env.3170050203>.

843 Pacyna, J.M., Pacyna, E.G., 2001. An assessment of global and regional emissions of trace metals to
844 the atmosphere from anthropogenic sources worldwide. *Environ. Rev.*, 9, 269-298.
845 <https://doi.org/10.1139/a01-012>.

846 Pacyna, E.G., Pacyna, J.M., Fudala, J., Strzelecka-Jastrzab, E., Hlawiczka, S., Panasiuk, D., Nitter, S.,
847 Pregger, T., Pfeiffer, H., Friedrich, R., 2007. Current and future emissions of selected heavy
848 metals to the atmosphere from anthropogenic sources in Europe. *Atmos. Environ.*, 41, 8557-
849 8566. <https://doi.org/10.1016/j.atmosenv.2007.07.040>.

850 Palleschi, S., Rossi, B., Armiento, G., Montereali, M.R., Nardi, E., Tagliani, S.M., Inglessis, M.,
851 Gianfagna, A., Silvestroni, L., 2018. Toxicity of the readily leachable fraction of urban PM2.5 to
852 human lung epithelial cells: role of soluble metals. *Chemosphere*, 196, 35–44.
853 <https://doi.org/10.1016/j.chemosphere.2017.12.147>.

854 Pan, Y.P., Wang, Y.S., 2014. Atmospheric wet and dry deposition of trace elements at ten sites in
855 Northern China. *Atmos. Chem. Phys. Discuss.*, 14, 20647-20676. [https://doi.org/10.5194/acp-](https://doi.org/10.5194/acp-15-951-2015)
856 [15-951-2015](https://doi.org/10.5194/acp-15-951-2015).

857 Pant, P., Harrison, R.M., 2012. Critical review of receptor modelling for particulate matter: A case
858 study of India. *Atmos. Environ.*, 49, 1-12. <https://doi.org/10.1016/j.atmosenv.2011.11.060>.

859 Pekney, N.J., Davidson, C.L., 2005. Determination of trace elements in ambient aerosol samples.
860 *Anal. Chim. Acta*, 540, 269-277. <https://doi.org/10.1016/j.aca.2005.03.065>.

861 Petäjä, T., Järvi, L., Kerminen, V.M., Ding, A.J., Sun, J.N., Nie, W., Kujansuu, J., Virkkula, A., Yang, X.Q.,
862 Fu, C.B., Zilitinkevich, S., Kulmala, M., 2016. Enhanced air pollution via aerosol-boundary layer
863 feedback in China. *Sci. Rep.*, 6, 18998. <https://doi.org/10.1038/srep18998> .

864 Pope, C.A., Brook, R.D., Burnett, R.T., Dockery, D.W., 2011. How is cardiovascular disease mortality
865 risk affected by duration and intensity of fine particulate matter exposure? An integration of
866 the epidemiologic evidence. *Air Qual. Atmos. Health*, 4, 5-14. [https://doi.org/10.1007/s11869-](https://doi.org/10.1007/s11869-010-0082-7)
867 [010-0082-7](https://doi.org/10.1007/s11869-010-0082-7).

868 Pryce, T.O., Baron, S., Bellina, B.H.M., Bellwood, P.S., Chang, N., Chattopadhyay, P., Dizon, E., Glover,
869 I.C., Hamilton, E., Higham, C.F.W., Kyaw, A.A, Laychour, V., Natapintu, S., Nguyen, V., Pautreau,
870 J.P., Pernicka, E., Pigott, V.C., Pollard, M., Pottier, C., Reinecke, A., Sayavongkhamdy, T.,
871 Souksavatdy, V., White, J., 2014. More questions than answers: the Southeast Asian Lead
872 Isotope Project 2009–2012. *J. Archaeol. Sci.*, 42, 273-294.
873 <https://doi.org/10.1016/j.jas.2013.08.024>.

874 Quang, T.N., Hue, N.T., Dat, M.V., Tran, L.K., Phi, T.H., Morawska, L., Thai, P.K., 2021. Motorcyclists
875 have much higher exposure to black carbon compared to other commuters in traffic of Hanoi,
876 Vietnam. *Atmos. Environ.*, 245, 118029. <https://doi.org/10.1016/j.atmosenv.2020.118029>.

877 Rafiee, A., Delgado-Saborit, J.M., D.Sly, P., Qu´emerai, B., Hashemi, F., Akbari, S., Hoseini, M., 2020.
878 Environmental chronic exposure to metals and effects on attention and executive function in
879 the general population. *Sci. Total Environ.*, 705, 135911.
880 <https://doi.org/10.1016/j.scitotenv.2019.135911>.

881 Rolph, G., Stein, A., Stunder, B., 2017. Real-time environmental applications and display system:
882 ready. *Environ. Model. Software*, 95, 210–228. <https://doi.org/10.1016/j.envsoft.2017.06.025>.

883 Roy, D., Singh, G., Seo, Y.C., 2019. Carcinogenic and non-carcinogenic risks from PM10-and PM2.5-
884 bound metals in a critically polluted coal mining area. *Atmos. Pollut. Res.*, 10, 1964–1975.
885 <https://doi.org/10.1016/j.apr.2019.09.002>.

886 Roy, S., Lam, Y.F., Hung, N.T., Chan, J.C.L., Fu, J.S., 2021. Development of 2015 Vietnam emission
887 inventory for power generation units. *Atmos. Environ.*, 247, 118042
888 <https://doi.org/10.1016/j.atmosenv.2020.118042>.

889 Sakundoo, P., Thonglua, T., Sangkham, S., Jirapornkul, C., Limmongkon, Y., Daduang, S., Tessiri, T.,
890 Rayubkul, J., Thongtip, S., Maneenin, N., Pimonsree, S., 2022. Human health risk assessment of
891 PM2.5-bound heavy metal of anthropogenic sources in the Khon Kaen Province of Northeast
892 Thailand. *Heyliyon*, 8, e09572. <https://doi.org/10.1016/j.heliyon.2022.e09572>.

893 Seinfeld, J.H., 2014. Tropospheric chemistry and composition: aerosols/particles. In: North, G.R., Pyle,
894 J., Zhang, F. (Eds.), Encyclopedia of Atmospheric Sciences, Second edition. Academic Press,
895 Oxford, pp. 182-187. <https://doi.org/10.1016/b978-0-12-382225-3.00438-2>.

896 Shen, Z.X., Cao, J.J., Arimoto, R., Zhang, R.J., Jie, D.M., Zhu, C.S., 2007. Chemical composition and
897 source characterization of spring aerosol over Horqin sand land in northeastern China. J.
898 Geophys. Res.: Atmos., 112, D14. <https://doi.org/10.1029/2006JD007991>.

899 Soetrismo, F.N., Delgado-Saborit, J.M., 2020. Chronic exposure to heavy metals from informal e-waste
900 recycling plants and children's attention, executive function and academic performance. Sci.
901 Total Environ., 717, 137099. <https://doi.org/10.1016/j.scitotenv.2020.137099>.

902 Stacey, J.S., Kramers, J.D., 1975. Approximation of terrestrial lead isotope evolution by a two-stage
903 model. Earth Planet. Sci. Lett., 26, 207-221. [https://doi.org/10.1016/0012-821X\(75\)90088-6](https://doi.org/10.1016/0012-821X(75)90088-6).

904 Stanaway, J.D., Afshin, A., Gakidou, E., Lim, S.S., Abate, D., Abate, K.H., Abbafati, C., et al., 2018.
905 Global, regional, and national comparative risk assessment of 84 behavioural, environmental
906 and occupational, and metabolic risks or clusters of risks for 195 countries and territories,
907 1990–2017: A systematic analysis for the Global Burden of Disease Study 2017. Lancet, 392,
908 1923-1994. [https://doi.org/10.1016/S0140-6736\(18\)32225-6](https://doi.org/10.1016/S0140-6736(18)32225-6).

909 Stein, A.F., Draxler, R.R., Rolph, G.D., Stunder, B.J.B., Cohen, M.D., Ngan, F., 2015. NOAA's HYSPLIT
910 atmospheric transport and dispersion modeling system. Bull. Am. Meteorol. Soc., 96, 2059–
911 2077. <https://doi.org/10.1175/BAMS-D-14-00110.1>.

912 Sun, R., Liu, G., Zheng, L., Chou, C.L., 2010. Geochemistry of trace elements in coals from the Zhuji
913 Mine, Huainan Coalfield, Anhui, China. Int. J. Coal Geol., 81, 81-96.
914 <https://doi.org/10.1016/j.coal.2009.12.001>.

915 Sun R., Sonke, J.E., Heimbürger, L.E., Belkin, H.E., Liu, G., Shome, D., Cukrowska, E., Liousse, C.,
916 Pokrovsky, O.S., Streets, D.G., 2014. Mercury Stable Isotope Signatures of World Coal Deposits
917 and Historical Coal Combustion Emissions. Environ. Sci. Technol., 48, 7660-7668.
918 <https://doi.org/10.1021/es501208a>.

919 Tang, Q., Liu, G., Zhou, C., Sun, R., 2013. Distribution of trace elements in feed coal and combustion
920 residues from two coal-fired power plants at Huainan, Anhui, China. *Fuel*, 107, 315-322.
921 <https://doi.org/10.1016/j.fuel.2013.01.009>.

922 Thorpe, A., Harrison, R.M., 2008. Sources and properties of non-exhaust particulate matter from road
923 traffic: a review. *Sci. Tot. Environ.*, 400, 270-282.
924 <https://doi.org/10.1016/j.scitotenv.2008.06.007>.

925 Tian, H., Zhou, J., Zhu, C., Zhao, D., Gao, J., Hao, J., He, M., Liu, K., Wang, K., Hua, S., 2014. A
926 Comprehensive Global Inventory of Atmospheric Antimony Emissions from Anthropogenic
927 Activities, 1995-2010. *Environ. Sci. Technol.*, 48, 10235-10241.
928 <https://doi.org/10.1021/es405817u>.

929 US. EPA, 1996. Method 3050B: Acid Digestion of Sediments, Sludges, and Soils, Revision 2.
930 Washington, DC.

931 US EPA, 2005. Aging and toxic response: issues relevant to risk assessment. EPA/600/P-03/004A.

932 US EPA, 2009. Risk assessment guidance for superfund. Volume I: Human health evaluation manual
933 (Part F, supplemental guidance for inhalation risk assessment). EPA-540-R-070-002, OSWER
934 9285.7-82.

935 US EPA, 2013. Reference method for the determination of fine particulate matter as PM 2.5 in the
936 atmosphere. 40 CRF 50.18. Appendix L.

937 Varrica, D., Bardelli, F., Dongarrà, G., Tamburo, E., 2013. Speciation of Sb in airborne particulate
938 matter, vehicle brake linings, and brake pad wear residues. *Atmos. Environ.*, 64, 18–24.
939 <https://doi.org/10.1016/j.atmosenv.2012.08.067>.

940 Vithanage, M., Bandara, P.C., Novo, L.A.B., Kumar, A., Ambade, B., Naveendrakumar, G., Ranagalage,
941 M., Magana-Arachchi, D.N., 2022. Deposition of trace metals associated with atmospheric
942 particulate matter: Environmental fate and health risk assessment. *Chemosphere*, 303,
943 135051. <https://doi.org/10.1016/j.chemosphere.2022.135051>.

944 Vo, L.H.T., Yoneda, M., Nghiem, T.D., Shimada, Y., Van, D.H., Nguyen, T.H.T., Nguyen, T.T., 2022.
945 Indoor PM0.1 and PM2.5 in Hanoi: Chemical characterization, source identification, and health
946 risk assessment. *Atmos. Pollut. Res.*, 13, 101324. <https://doi.org/10.1016/j.apr.2022.101324>.

947 Vuong, Q.T., Bac, V.T., Thang, P.Q., Park, M.K., Choi, S.D., 2023. Trace element characterization and
948 source identification of particulate matter of different sizes in Hanoi, Vietnam. *Urban Climate*,
949 48, 101408. <https://doi.org/10.1016/j.uclim.2023.101408>.

950 Wang, X., Dickinson, R.E., Su, L., Zhou, C., Wang, K., 2018. PM2.5 pollution in China and how it has
951 been exacerbated by terrain and meteorological conditions. *Bull. Am. Meteorol. Soc.*, 99, 105-
952 119. <https://doi:10.1175/BAMS-D-16-0301.1>.

953 Wedepohl, K.H., 1995. The composition of the continental crust. *Geochim. Cosmochim. Acta*, 59, 217-
954 1232. [http://dx.doi.org/10.1016/0016-7037\(95\)00038-2](http://dx.doi.org/10.1016/0016-7037(95)00038-2).

955 Wilcox, E.M., Thomas, R.M., Praveen, P.S., Ramanathan, V., 2016. Black carbon solar absorption
956 suppresses turbulence in the atmospheric boundary layer. *PNAS*, 113, 11794-11799.
957 <https://doi.org/10.1073/pnas.1525746113>.

958 Witkowska, D., Słowik, J., Chilicka, K., 2021. Heavy metals and human health: possible exposure
959 pathways and the competition for protein binding sites. *Molecules*, 26, 6060. doi:
960 10.3390/molecules26196060. <https://doi.org/10.3390/molecules26196060>.

961 World Health Organization, 2021. WHO global air quality guidelines: particulate matter (PM2.5 and
962 PM10), ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide. World Health
963 Organization. <https://apps.who.int/iris/handle/10665/345329>.

964

965 Xing, Y.F., Xu, Y.H., Shi, M.H., Lian, Y.X., 2016. The impact of PM2.5 on the human respiratory system.
966 *J. Thorac. Dis.*, 8, E69-E74. <https://doi.org/10.3978/j.issn.2072-1439.2016.01.19>.

967 Xu, J., Bergin, M.H., Greenwald, R., Schauer, J.J., Shafer, M.M., Jaffrezo, J.L., Aymoz, G., 2004a.
968 Aerosol chemical, physical, and radiative characteristics near a desert source region of

969 northwest China during ACE-Asia. *J. Geophys. Res.: Atmos.*, 109, D19.
970 <https://doi.org/10.1029/2003JD004239>.

971 Xu, M., Yan, R., Zheng, C., Qiao, Y., Han, J., Sheng, C., 2004b. Status of trace element emission in a
972 coal combustion process: a review. *Fuel Process. Technol.*, 85, 215-237.
973 [https://doi.org/10.1016/S0378-3820\(03\)00174-7](https://doi.org/10.1016/S0378-3820(03)00174-7).

974 Xu, H.M., He, K.L., Feng, R., Shen, Z.X., Cao, J.J., Liu, S.X., Ho, K.F., Huang, R.J., Guinot, B., Wang, Q.Y.,
975 Zhou, J.M., Shen, M.X., Xiao, S., Zhou, B.H., Sonke, J.E., 2020. Metallic elements and Pb
976 isotopes in PM_{2.5} in three Chinese typical megacities: spatial distribution and source
977 apportionment. *Environ. Sci.*, 22, 1718-1730. <https://doi.org/10.1039/d0em00174k>.

978 Yang, X., Wang, T., Xia, M., Gao, X., Li, Q., Zhang, N., Yuan, G., Lee, S., Wang, X., Xue, L., Yang, L.,
979 Wang, W., 2018. Abundance and origin of fine particulate chloride in continental China. *Sci.*
980 *Total Environ.*, 624, 1041-1051. <https://doi.org/10.1016/j.scitotenv.2017.12.205>.

981 Yao, P.H., Shyu, G.S., Chang, Y.F., Chou, Y.C., Shen, C.C., Chou, C.S., Chang, T.K., 2015. Lead isotope
982 characterization of petroleum fuels in Taipei, Taiwan. *Int. J. Environ. Res. Public Health*, 12,
983 4602-4616. <https://doi.org/10.3390/ijerph120504602>.

984 Yuan, Q., Qi, B., Hu, D., Wang, J., Zhang, J., Yang, H., Zhang, S., Liu, L., Xu, L., Li, W., 2021.
985 Spatiotemporal variations and reduction of air pollutants during the COVID-19 pandemic in a
986 megacity of Yangtze River Delta in China. *Sci. Total Environ.*, 751, 141820
987 <https://doi.org/10.1016/j.scitotenv.2020.141820>.

988 Zhang, J., Ren, D., Zhu, Y., Chou, C.L., Zeng, R., Zheng, B., 2004. Mineral matter and potentially
989 hazardous trace elements in coal from Qianxi Fault Depression Area in southwestern Guizhou,
990 China. *Int. J. Coal Geol.*, 57, 49-61. <https://doi.org/10.1016/j.coal.2003.07.001>.

991 Zhang, X., Eto, Y., Aikawa, M., 2021. Risk assessment and management of PM_{2.5}-bound heavy metals
992 in the urban area of Kitakyushu, Japan. *Sci. Tot. Environ.*, 795, 148748.
993 <https://doi.org/10.1016/j.scitotenv.2021.148748>.

994 Zhao, Z.Q., Zhang, W., Li, X.D., Yang, Z., Zheng, H.Y., Ding, H., Wang, Q.L., Xiao, J., Fu, P.Q., 2015.
995 Atmospheric lead in urban Guiyang, Southwest China: Isotopic source signatures. *Atmos.*
996 *Environ.*, 115, 163-169. <https://doi.org/10.1016/j.atmosenv.2015.05.049>.

997 Zheng, J., Tan, M., Yasuyuki, S., Atsushi, T., Li, Y., Zhang, G., Zhang, Y., Shan, Z., 2004. Characteristics
998 of lead isotope ratios and elemental concentrations in PM₁₀ fraction of airborne particulate
999 matter in Shanghai after the phase-out lead gasoline. *Atmos. Environ.*, 38, 1191-1200.
1000 <https://doi.org/10.1016/j.atmosenv.2003.11.004>.

1001 Zhou, B., Liu, D., Yan, W., 2021. A Simple New Method for Calculating Precipitation Scavenging Effect
1002 on Particulate Matter: Based on Five-Year Data in Eastern China. *Atmos.*, 12, 759,
1003 <https://doi.org/10.3390/atmos12060759>.

1004 Zhou, X., Xie, M., Zhao, M., Wang, Y., Luo, J., Lu, S., Li, J., Liu, Q., 2023. Pollution characteristics and
1005 human health risks of PM_{2.5}-bound heavy metals: a 3-year observation in Suzhou, China.
1006 *Environ. Geochem. Health*, 45, 5145-5162. <https://doi.org/10.1007/s10653-023-01568-x>.

1007 Zhu, B.Q., Chen, Y.W., Peng, J.H., 2001. Lead isotope geochemistry of the urban environment in the
1008 Pearl River Delta. *Appl. Geochem.*, 16, 409-417. [https://doi.org/10.1016/S0883-](https://doi.org/10.1016/S0883-2927(00)00047-0)
1009 [2927\(00\)00047-0](https://doi.org/10.1016/S0883-2927(00)00047-0).

1010 Zhuang, X.G., Querol, X., Zeng, R.S., Xu, W.D., Alastuey, A., Lopez-Soler, A., Plana, F., 2000. Mineralogy
1011 and geochemistry of coal from the Liupanshui mining district, Guizhou, South China. *Int. J. Coal*
1012 *Geol.*, 45, 21-37. [https://doi.org/10.1016/S0166-5162\(00\)00019-7](https://doi.org/10.1016/S0166-5162(00)00019-7).

1013

1014

1015

1016 **Tables**

1017 **Table 1.** Variations (Mean, Min and Max) of PM2.5-bound MM concentrations (ng m⁻³) during the sampling period (November 2019 to December 2020).

1018 Mean and standard deviation (σ) include all samples of this study (n = 57).

	Ag	Al	As	Cd	Co	Cr	Cu	Fe	Hg	Mn	Ni	Sb	Sn	Pb	Ti	U	V	Zn
Mean	0.285	349	3.97	1.82	0.153	3.31	56.8	295	0.190	32.0	1.34	4.81	3.38	65.7	26.3	0.039	1.20	444
σ	0.467	271	2.80	1.43	0.090	5.31	90.9	198	0.134	43.9	1.31	3.83	2.31	71.6	19.2	0.031	0.700	545
Min	0.012	15.4	0.674	0.179	0.028	0.412	2.28	70.9	0.010	2.83	0.018	0.646	0.784	3.29	4.29	0.001	0.137	22.5
Max	3.47	1427	11.2	5.95	0.477	40.5	641	1033	0.576	275	7.01	16.9	10.8	392	98.2	0.142	3.95	2054

1019

1020

1021

1022 **Table 2.** Pb isotopic compositions in estimated natural soil, urban aerosol and relics from China and
 1023 southeast Asian.

			$^{207}\text{Pb}/^{206}\text{Pb}$	$^{208}\text{Pb}/^{206}\text{Pb}$
<i>Estimated natural soil (Bing-Quan et al., 2002)</i>				
Northeast China	min		0.8673	2.1084
Northeast China	max		0.8511	2.0800
North China	min		0.9615	2.2817
North China	max		0.8621	2.1457
Indochina	min		0.8410	2.0698
Indochina	max		0.8278	2.0613
<i>Aerosol from urban cities (Bollhöfer and Rosman, 2000)</i>				
Vietnam	Heni		0.8569	2.1020
Vietnam	Huzhimin		0.8658	2.1039
Thailand	Bangkok		0.8873	2.1331
Malaysia	Kuala Lumpur		0.8764	2.1122
Indonesia	Bandung		0.9116	2.1568
<i>Neolithic relics from southeast Asia (Pryce et al., 2014)</i>				
Thailand	Khao SaKaeo	400-200 BC	0.7695	2.0063
Thailand	Non Pa Wai	1000-300 BC	0.7785	1.8921
Thailand	Phu Lon	1000 BC	0.7877	1.9534
Vietnam	Lang Nhon	NA	0.7958	2.0531

1024 NA: not available

1025

1026

1027 **Table 3.** Estimation of human health risk from PM2.5 inhalation in Hanoi: detailed indices (HQ_i and
 1028 CR_i) per element and cumulative indices (HI, TCR).

	HQ_i	CR_i
Ag	NA	NA
Al	$7.4 \cdot 10^{-5}$	NA
As	$2.9 \cdot 10^{-4}$	$6.3 \cdot 10^{-6}$
Cd	$2.1 \cdot 10^{-4}$	$3.0 \cdot 10^{-6}$
Co	$2.7 \cdot 10^{-5}$	$5.0 \cdot 10^{-7}$
Cr	$3.1 \cdot 10^{-2}$	$8.8 \cdot 10^{-5}$
Cu	$1.5 \cdot 10^{-3}$	NA
Fe	NA	NA
Hg	$6.7 \cdot 10^{-6}$	NA
Mn	$6.6 \cdot 10^{-4}$	NA
Ni	$9.5 \cdot 10^{-5}$	$1.2 \cdot 10^{-7}$
Pb	$2.2 \cdot 10^{-2}$	$3.2 \cdot 10^{-7}$
Sb	$1.8 \cdot 10^{-2}$	NA
Sn	NA	NA
Ti	$2.8 \cdot 10^{-4}$	NA
U	$1.1 \cdot 10^{-6}$	NA
V	$1.8 \cdot 10^{-4}$	$3.7 \cdot 10^{-6}$
Zn	$1.7 \cdot 10^{-3}$	NA
	HI	TCR
Total	$7.6 \cdot 10^{-2}$	$1.0 \cdot 10^{-4}$

1029 NA: not available

1030

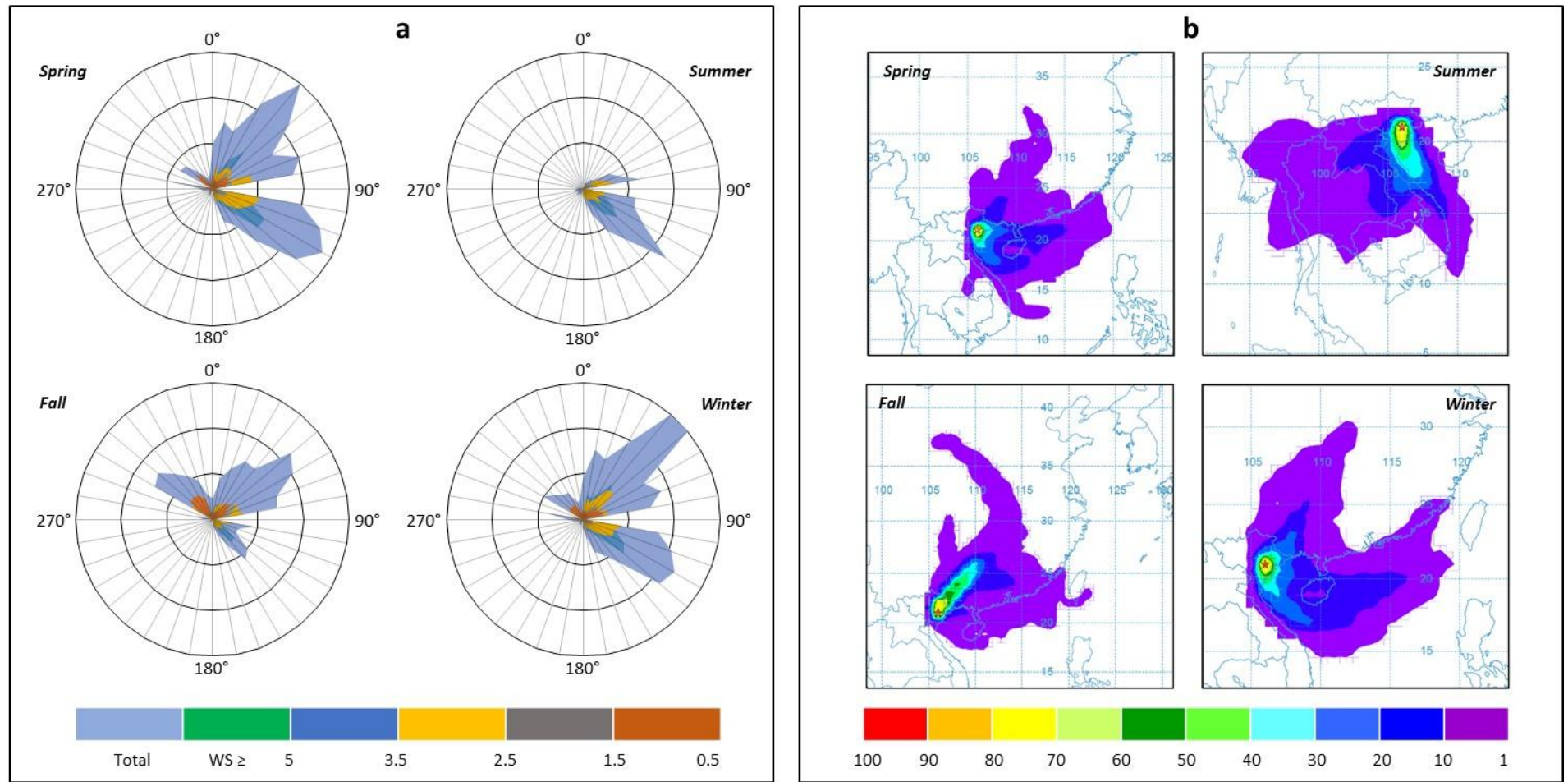


Figure 1: Localisation of sampling site

1031

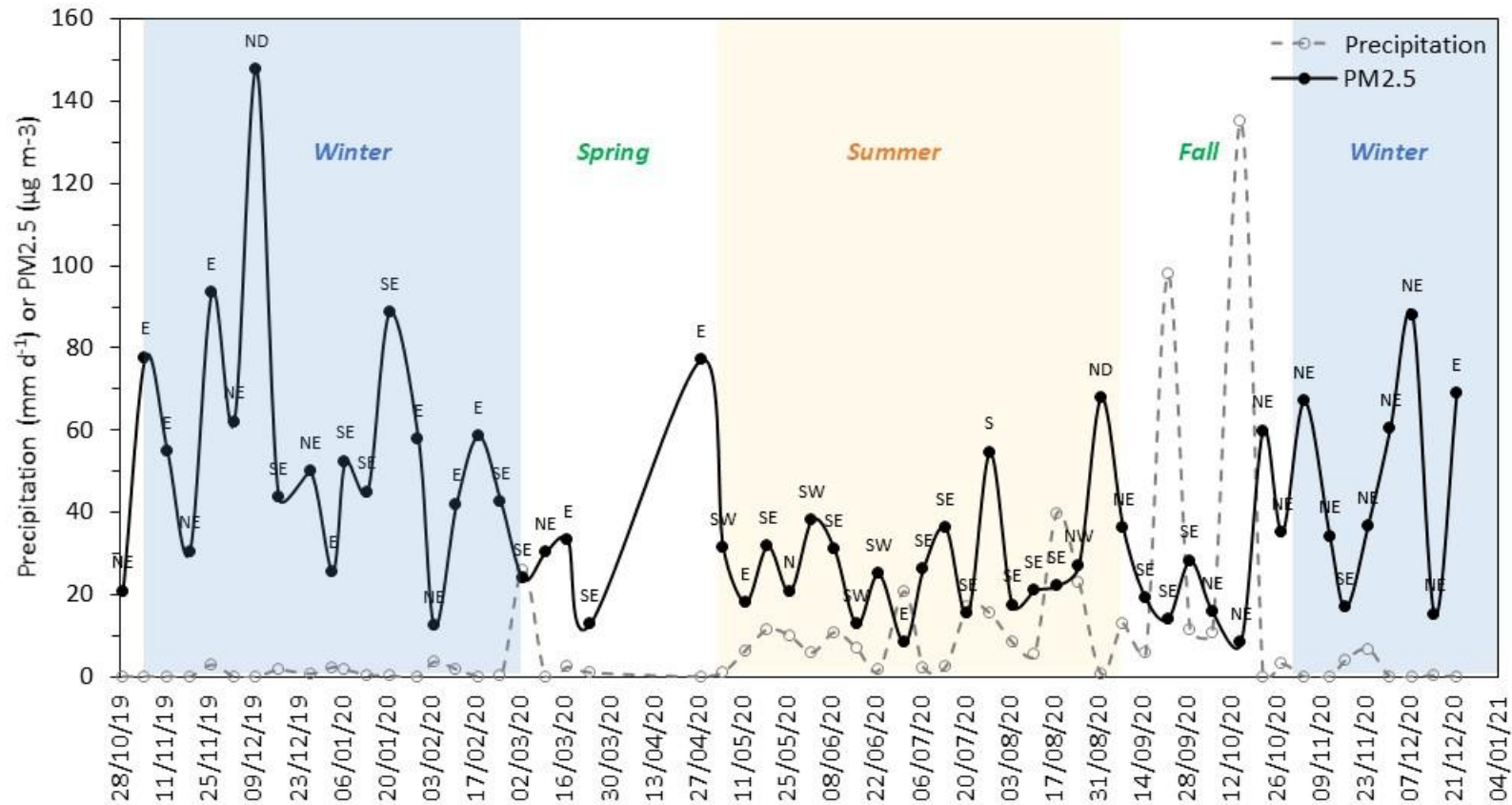
1032

1033



1034

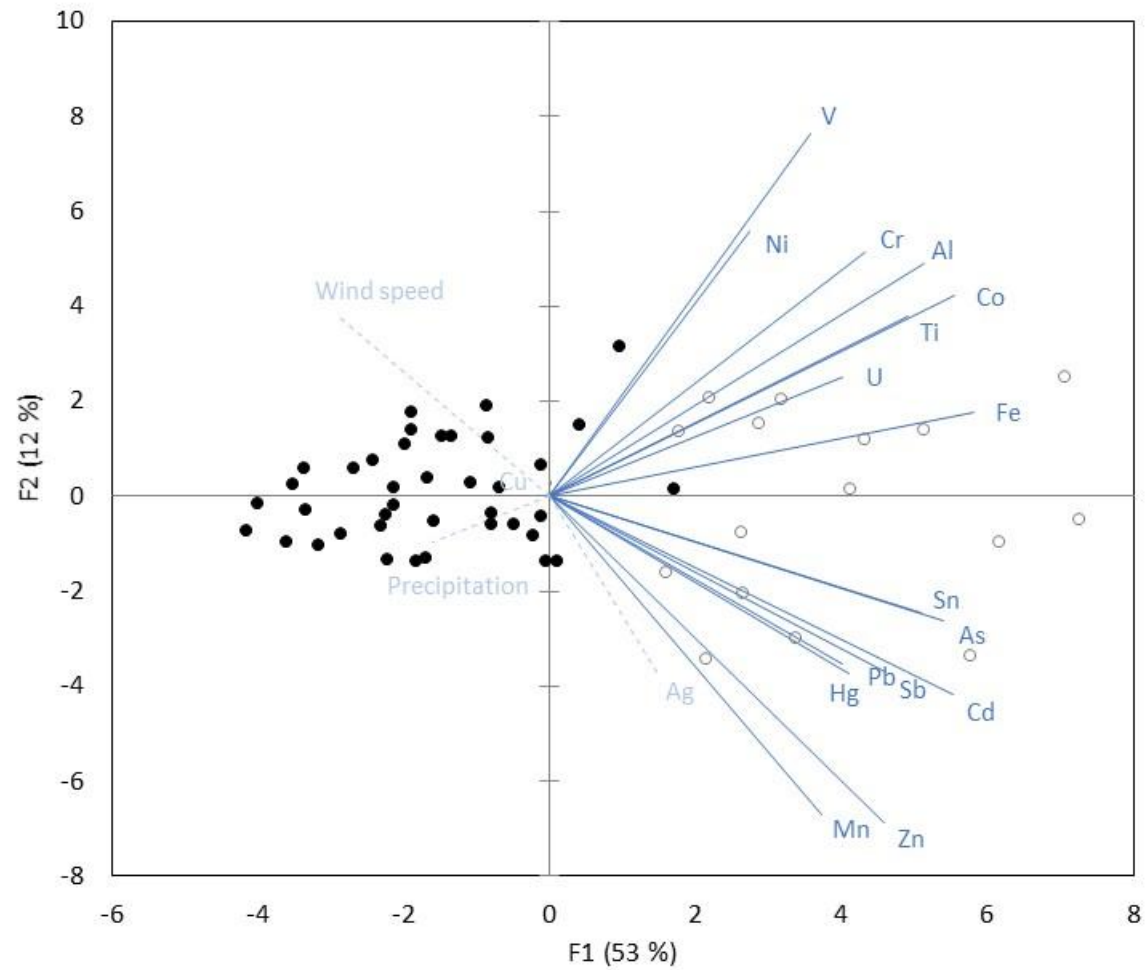
1035 **Figure 2:** Wind roses (a) and trajectory frequencies (b) grouped by seasonal variations during the sampling period. Colour scales of wind speed (WS) and
 1036 trajectory frequencies are expressed in m s⁻¹ and %, respectively (NOAA HYSPLIT model, <https://www.ready.noaa.gov/HYSPLIT.php>).



1037

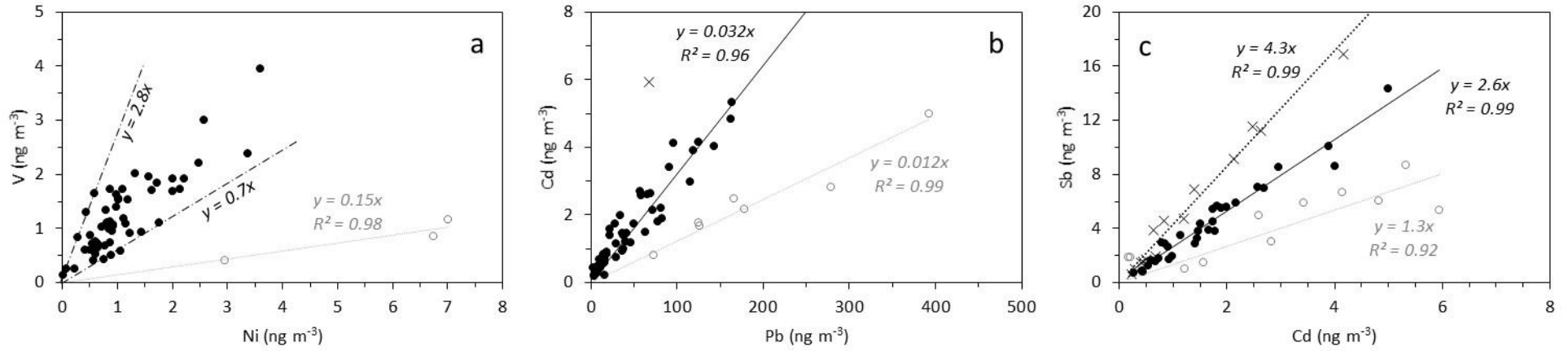
1038 **Figure 3:** Seasonal variations in PM2.5 concentrations (μg m⁻³) and precipitation (mm d⁻¹). Abbreviations indicate the wind direction: Northeast (NE), East (E),
 1039 Southeast (SE), Southwest (SW), Northwest (NW) and not defined (ND)

1040



1041

1042 **Figure 4:** Statistical analyses (PCA-HCA) of MM concentrations in PM_{2.5} during the sampling period (November 2019 to December 2020). The contribution
 1043 of the active and illustrative variables on axes F1 and F2 are presented in solid and dotted lines, respectively. The distribution of samples in clusters C1
 1044 and C2 are presented in solid and open circles, respectively.

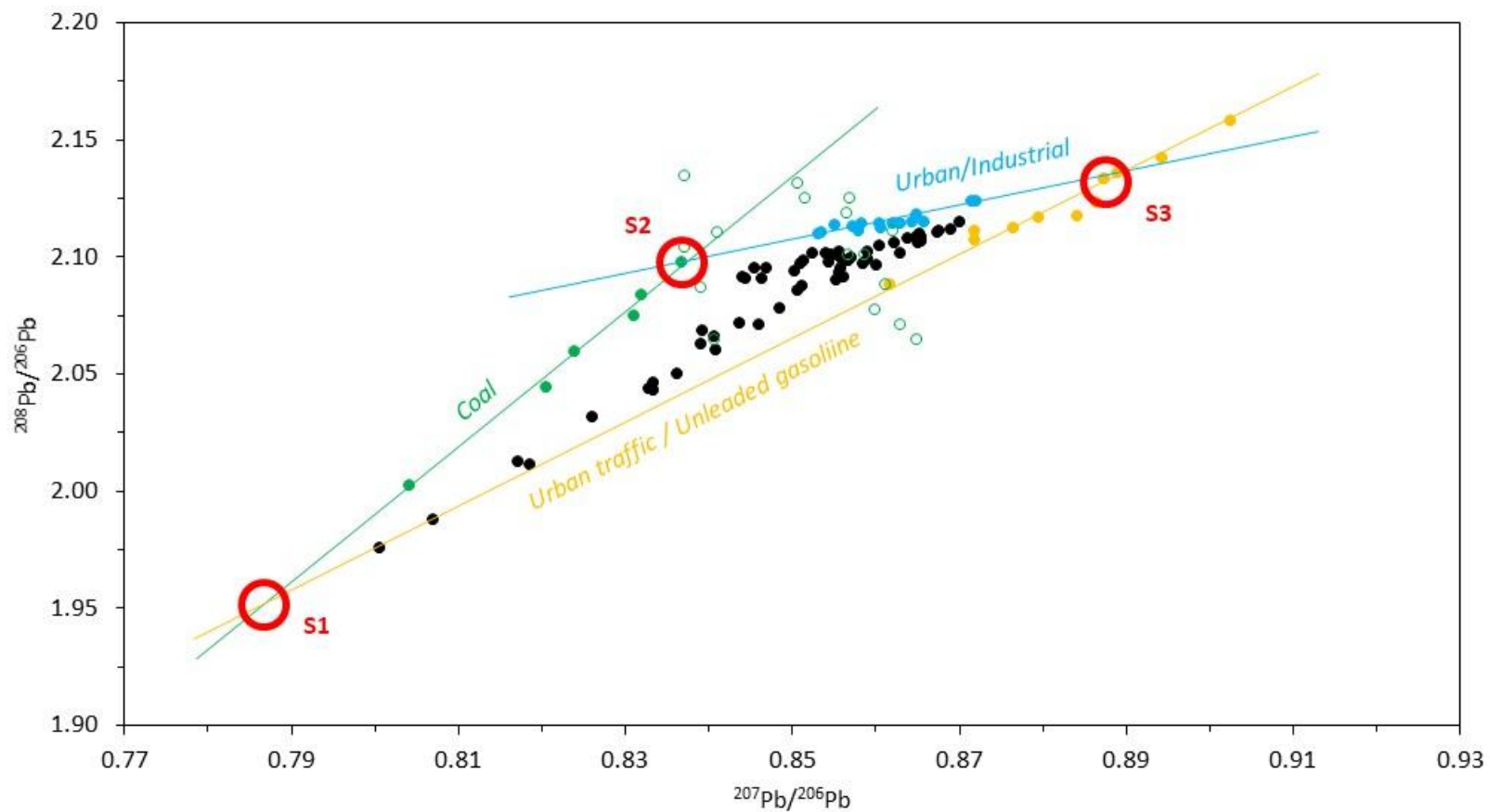


1045

1046

Figure 5: Elementary ratios in PM_{2.5} collected during the sampling period (November 2019 to December 2020): a) V/Ni, b) Cd/Pb, c) Sb/Cd and d) Al/Ti

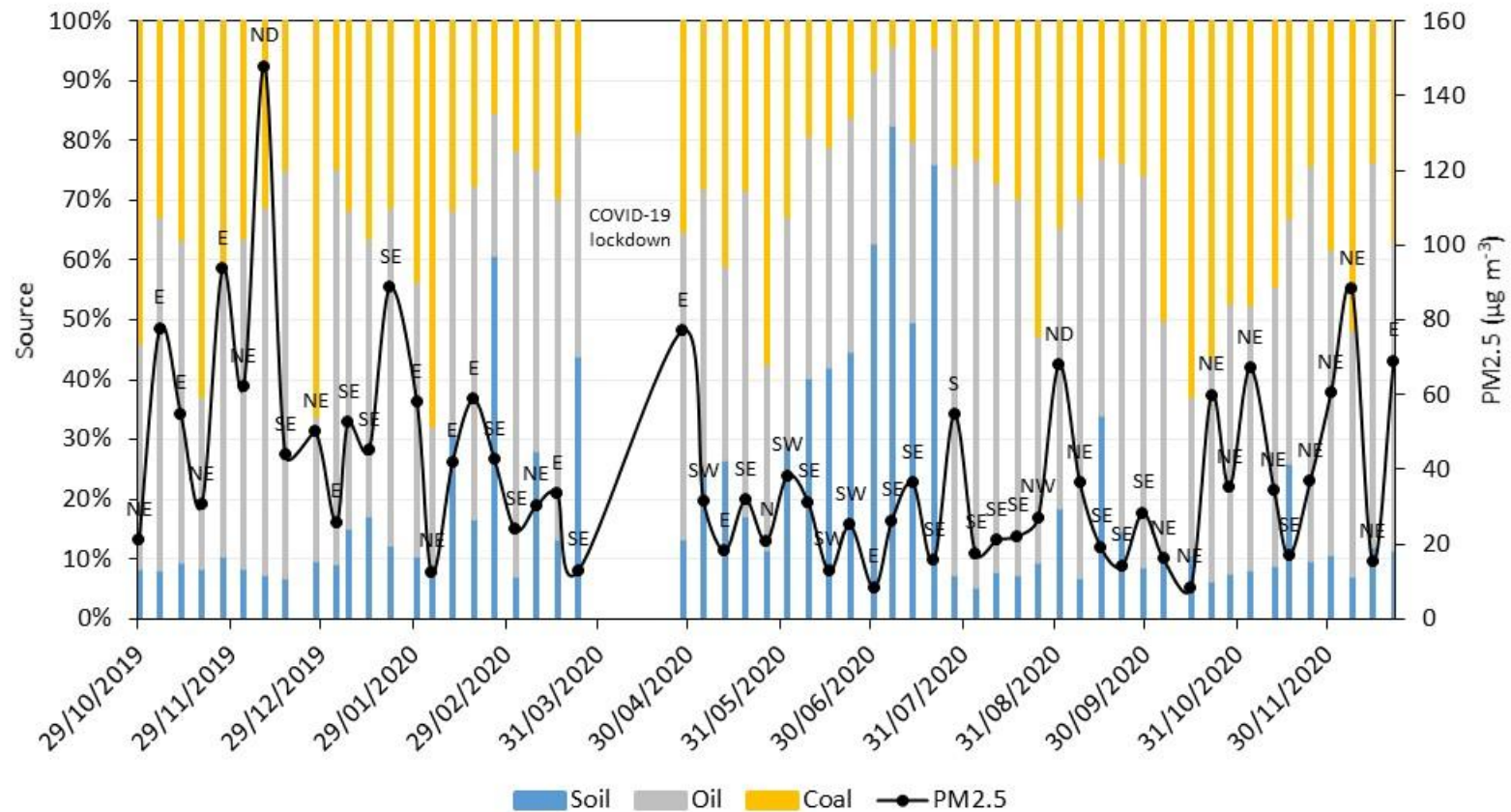
1047



1048

1049 **Figure 6:** Pb isotopic ratios ($^{208}\text{Pb}/^{206}\text{Pb}$ vs $^{207}\text{Pb}/^{206}\text{Pb}$) in PM_{2.5} collected in Hanoi during the sampling period (in black). The coloured data comes from the
 1050 literature, in green for Chinese coal mines (solid circles come from southern provinces), in blue for Chinese urban and industrial activities and in
 1051 orange for petroleum-related activities (*e.g.*, from road traffic in Asian megacities, unleaded Chinese gasoline and Russian oil fields).

1052



1053

1054 **Figure 7:** Contribution of Pb sources from 'natural' soil, 'oil' and coal to PM2.5 during the sampling period (November 2019 to December 2020).

1055 Abbreviations indicate the wind direction: Northeast (NE), East (E), Southeast (SE), Southwest (SW), Northwest (NW) and not defined (ND)

1056